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# Sand quarry wetlands provide high-quality habitat for native amphibians

**Michael Sievers**[1,2]

[1]School of BioSciences, University of Melbourne, Parkville, Victoria, 3010, Australia
[2]School of Ecosystem and Forest Sciences, University of Melbourne, Parkville, Victoria, 3010, Australia

*Correspondence to:* Michael Sievers (msievers@student.unimelb.edu.au)

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**Abstract.** Anthropogenic disturbances to habitats influence the fitness of individual animals, the abundance of their populations, and the composition of their communities. Wetlands in particular are frequently degraded and destroyed, impacting the animals that inhabit these important ecosystems. The creation of wetlands during and following sand extraction processes is inevitable, and thus, sand quarries have the potential to support aquatic animals. To determine how amphibians utilise these wetlands, I conducted nocturnal call surveys at wetlands within the Kables Sands quarry, New South Wales, Australia, and within surrounding reference wetlands, and quantified levels of developmental instability (DI) as a proxy for fitness. Whilst quarry and reference wetlands were largely similar in terms of environmental characteristics, quarry wetlands consistently harboured more amphibian species and individuals. Using unsigned asymmetry as a measure of DI, frogs from the quarry sites exhibited significantly lower levels of DI compared to reference wetlands, indicating that quarry wetlands may be comparatively higher quality. Levels of DI within quarry wetlands also compared favourably to data from healthy frog populations extracted from the literature. Further enhancing the suitability of quarry wetlands would require minimal effort, with potentially significant increases in local and regional biodiversity. Documenting species presence and quantifying individual fitness by measuring limb lengths is an economically and logistically feasible method to assess the health of quarry wetlands. Overall, the methods outlined here provide a powerful, yet simple, tool to assess the overall health and suitability of quarry wetlands that could be easily adopted at quarries throughout the world.

## 1 Introduction

Humans are altering natural environments at unprecedented rates, with significant negative impacts for animals (MEA, 2005; Pereira et al., 2010). Wetlands in particular harbour highly diverse biological communities and provide multiple ecosystem services, yet are frequently degraded and destroyed, with over 50 % of global wetland surface area lost during the last century (Mitsch and Gosselink, 2007; Zedler and Kercher, 2005). Consequently, declines of wetland-dependent species are some of the greatest recorded (MEA, 2005). Concurrent with these losses, wetlands are being created to harness the ecosystem services they provide, such as those constructed in urban areas to treat storm water, and settlement dams created in mining areas to store and treat processed materials (Hammer, 1989; Odum, 2016).

Although typically not designed to support and conserve wildlife, these artificial wetlands regularly attract animals, as they superficially resemble natural wetlands and contain the cues used by animals when selecting habitats (often termed secondary wetlands; Dolny and Harabis, 2012). Therefore, secondary wetlands may provide critical habitat, enhancing landscape-level connectivity and promoting biodiversity and population persistence (Bendell-Young et al., 2000; Scheffers and Paszkowski, 2013; Thiere et al., 2009). For some organisms, properly managed secondary wetlands may provide habitat superior to that of natural wetlands. For example, Dolny and Harabis (2012) observed more than twice as many dragonfly species in mine subsidence pools than in reference wetlands, suggesting that this was due to enhanced environmental heterogeneity resulting from abiotic succession processes occurring as a direct consequence of mining. Given this potential, there has been considerable interest in simultaneously satisfying industrial needs and promoting wildlife within secondary wetlands (i.e. multi-objective management; Benyamine et al., 2004).

However, since secondary wetlands are often designed to store contaminated water, or are in areas prone to contamination and human interference, animal inhabitants may experience compromised fitness (e.g. reduced survival and reproduction; Dods et al., 2005; Laposata and Dunson, 2000). Therefore, despite being conceptually appealing, multi-objective management may prove problematic if individual fitness within secondary wetlands is impacted, and particularly if individual fitness impacts scale up to affect population persistence.

Quantifying community composition, population sizes and individual fitness will assist management efforts to maintain and enhance the conservation potential of secondary wetlands. In particular, utilising fast, cheap and non-lethal methods of estimating individual fitness is preferable under certain circumstances. Quantifying developmental instability (DI) is a promising method for estimating the fitness of individuals and provides an indirect proxy for habitat quality (Alford et al., 1999; Burghelea et al., 2013; Tracy et al., 1995). Developmental stability occurs when genotypes repeatedly produce the same phenotype under the same environmental conditions during development (Tracy, et al., 1995; Zakharov, 1992), so levels of deviation from stability (i.e. DI) provide information on environmental quality and individual health. DI is calculated by measuring bilateral structures such as the length of limbs and quantifying disparity from symmetry. Studies have shown that DI increases as health decreases, and that amphibian population declines have preceded periods of increasing DI (Alford et al., 2007). Therefore, DI may be a valuable, non-lethal early warning indicator of the impact of poor-quality wetlands.

Determining the suitability of secondary wetlands for currently threatened taxa in particular should be at the forefront of conservation science. Amphibian populations are currently experiencing dramatic declines around the world, with approximately 40 % of species facing the threat of extinction (Monastersky, 2014; Whitfield et al., 2007). Amphibians are often considered particularly sensitive to environmental contamination due to their physiology and biphasic life cycle (Wake and Vredenburg, 2008). Although numerous factors have been implicated in current population declines, such as climate change, disease and pollution, habitat loss clearly stands out as one of the main threats facing amphibians (Cushman, 2006). Consequently, secondary wetlands such those within sand quarries may provide vitally important habitat. Relative to other mining industries which may release materials into wetlands that are highly toxic to wildlife, such as fossil fuel mining, sand quarry wetlands may be more suitable due to the comparably lower environmental impact of this industry (Anderson and Arruda, 2006; Rowe et al., 1996). However, there are few empirical studies investigating how sand quarry wetlands function as habitat for amphibians or indeed any other animal groups (although see Catchpole and Tydeman, 1975; Eversham et al., 1996; Spencer and Griffith, 2012).

Here, I determine how quarry wetlands within the Kables Sands Quarry, Australia, function as habitat for local amphibians, and investigate the factors that may be influencing amphibian populations and communities. To do this, I measured environmental variables and quantified amphibian population density, species richness and levels of developmental instability (DI) within sand quarry wetlands, and compared findings to reference wetlands and previously published studies.

## 2 Methods

### 2.1 Study site selection and description

The Kables Sands Quarry (33°27′27.51″ S, 150°14′26.04″ E) is located in Clarence, New South Wales, Australia (Fig. 1). The extraction of sandstone and production of sand began here 60 years ago, with current production at approximately 350 000 tonnes per annum. Mining activities have inadvertently led to the formation of wetlands filled with groundwater, and other wetlands have been intentionally created for sand washing and other production purposes. The four accessible wetlands within the quarry were surveyed: the settlement dam (Q1), quarry swamp (Q2), tailings storage cell (Q3) and the reservoir (Q4; Fig. 1). Quarry wetlands are surrounded by terrestrial areas currently undergoing rehabilitation, and are between 10 and 50 years old (see Table 1). The adjacent suburb, Lithgow (33°28′51.19″ S, 150°09′26.67″ E), was searched using high-resolution aerial imagery to locate reference wetlands. Those that were accessible were selected for surveying, with highly isolated sites within densely forested or fenced areas excluded due to logistical and safety issues. This limited reference sites to five wetlands within suburban areas: Geordie St Wetland (R1), Laidley St Wetland (R2), Vale of Clwydd Wetland (R3), Lake Pillans Wetland 1 (R4) and Lake Pillans Wetland 2 (R5; Fig. 1).

### 2.2 Local habitat variables

Wetland area and perimeter were calculated using online spatial analysis tools available at www.nearmap.com.au. During surveys, I measured variables that may influence amphibian detection rates: date, time, cloud cover, wind speed and current rainfall. In addition, approximately 100 mL of wetland water was collected from three equidistant points around the pond perimeter (1 m from the shoreline and at a depth of 5–10 cm), mixed and tested for water temperature, conductivity and pH using a hand-held electronic meter (WP-81 meter; TPS, Brisbane, Australia). Local air temperature, humid-

**Table 1.** Site characteristics including wetland area, perimeter and age, water quality data and the percentage cover of vegetation in various regions within wetlands. Reference wetlands (ref) – R1 is Geordie St Wetland; R2 is Laidley St Wetland; R3 is Vale of Clwydd Wetland; R4 is Lake Pillans Wetland 1; R5 is Lake Pillans Wetland 2. Quarry wetlands (quarry) – Q1 is settlement dam; Q2 is quarry swamp; Q3 is tailings storage cell; Q4 is reservoir.

| | R1 | R2 | R3 | R4 | R5 | Q1 | Q2 | Q3 | Q4 |
|---|---|---|---|---|---|---|---|---|---|
| Site type | Ref | Ref | Ref | Ref | Ref | Quarry | Quarry | Quarry | Quarry |
| Area ($m^2$) | 1667 | 461 | 2678 | 3888 | 2305 | 12 420 | 1831 | 8588 | 3760 |
| Perimeter (m) | 272 | 134 | 514 | 369 | 199 | 453 | 225 | 391 | 507 |
| Approx. age (yr) | 100+* | 100+* | 100+* | 105 | 105 | 10 | 10 | 30 | 50 |
| Survey period 1 (30/03/16–01/04/16) | | | | | | | | | |
| Water temp (°C) | 15.7 | 15.8 | 15.0 | 17.9 | 17.2 | 16.5 | – | 17.5 | 17.0 |
| pH | 7.6 | 7.8 | 7.2 | 7.6 | 7.4 | 7.3 | – | 6.1 | 6.2 |
| Conductivity (µS) | 133.2 | 88.1 | 157.0 | 147.2 | 166.6 | 30.1 | – | 25.4 | 20.8 |
| Survey period 2 (20/07/16–21/07/16) | | | | | | | | | |
| Water temp (°C) | 10.1 | 10.4 | 10.9 | 11.0 | 11.7 | 10.8 | – | 10.1 | 10.1 |
| pH | 7.0 | 7.5 | 7.1 | 7.6 | 7.5 | 6.0 | – | 5.8 | 5.1 |
| Conductivity (µS) | 176.4 | 188.8 | 199.2 | 138.5 | 148.7 | 20.4 | – | 16.5 | 23.8 |
| Percentage vegetation | | | | | | | | | |
| Submerged (%) | 0 | 20 | 60 | 10 | 0 | 0 | – | 0 | 0 |
| Emergent (%) | 10 | 20 | 90 | 20 | 80 | 10 | 100 | 3 | 30 |
| Fringing (%) | 80 | 100 | 100 | 100 | 100 | 5 | 100 | 30 | 90 |
| Floating (%) | 0 | 0 | 20 | 0 | 0 | 0 | – | 0 | 0 |

* Age of wetlands could not be precisely determined. However, all are in-line wetlands connected to long-established creeks or manmade channels formed or constructed, respectively, over 100 years ago. Note that water quality variables could not be measured within the swamp due to safety concerns over entering this site.

ity and total rainfall during the survey period were extracted from the Bureau of Meteorology website (www.bom.gov.au). I visually estimated the percentage cover of vegetation in the fringing, emergent, submerged and floating regions of each wetland (Parris, 2006).

## 2.3 Amphibian surveys

Each wetland was surveyed for amphibians 3–5 times over two distinct temporal periods: during March–April and July 2016. The sequence in which wetlands were visited was randomised and stratified according to wetland proximity, since wetlands occurred in clusters (i.e. quarry wetlands). Adopting standard nocturnal search methods (see Parris et al., 1999), I listened for the advertisement calls of male frogs at a site for a minimum of 15 min, and then searched the wetland and surrounding banks and vegetation with a headlamp for a minimum of 30 min. Frogs were captured, identified to species level, weighed and measured (snout-vent and limb lengths). Species-specific abundance was estimated by categorising advertisement call intensity according to the index provided by Pope and colleagues (2000): (0) no individuals calling; (1) individual(s) can be counted with calls not overlapping; (2) calls of < 15 individuals can be distinguished, but there is some overlapping; and (3) > 15 individuals are calling.

## 2.4 Developmental instability

To calculate DI, I examined the total unsigned asymmetry of each collected frog (i.e. the difference between the left and right forelimbs, plus the corresponding value for the hindlimbs) using the blind protocol method developed by Alford et al. (1999). Additionally, informal comparisons with values extracted from Alford et al. (2007) were made in order to consider these DI data from a more biologically relevant perspective. These authors provided temporal trends in DI, so I calculated the mean DI over this period for control and impact groups. Although this provides only a simplified quantification of levels of DI in the different populations, it nonetheless provides a relevant comparator with which to compare measurements here.

## 2.5 Data analysis

I used two general linear multivariate models to examine the influence of environmental and local habitat characteristics on species richness and the indexed abundance of *C. signifera*. The first model consisted of more temporally stable factors: treatment (quarry or reference wetlands), wetland perimeter (log-transformed) and a vegetation index (summed proportion of emergent and fringing vegetation) were all fitted as fixed effects. The second model included water tem-

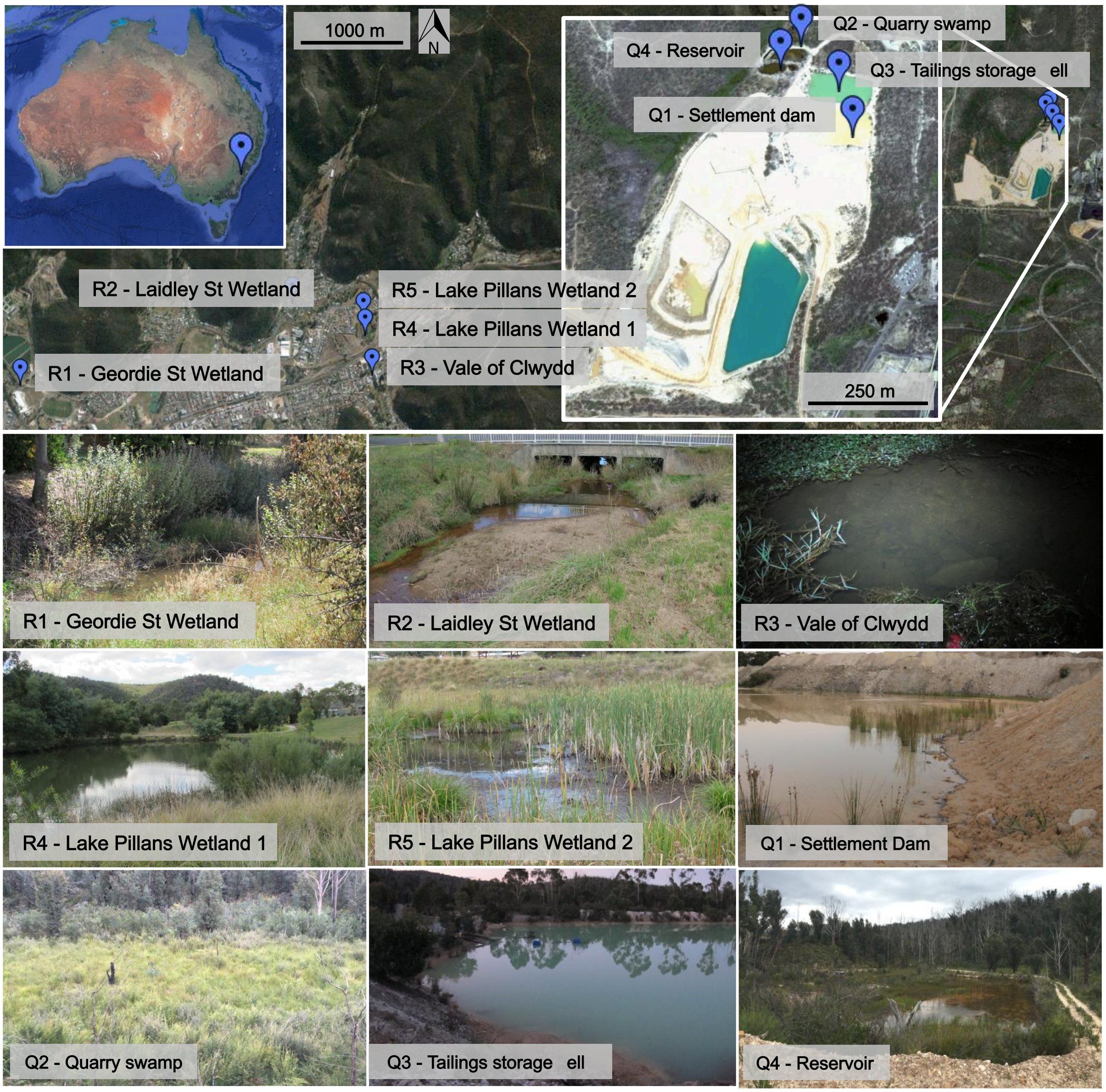


**Figure 1.** Map of the study area (the suburbs of Lithgow and Clarence, New South Wales, Australia), with the Kables Sands Quarry expanded inset, showing site locations, and photographs of the nine wetlands surveyed. Reference wetlands: R1 is Geordie St Wetland, R2 is Laidley St Wetland, and R3 is Vale of Clwydd Wetland, R4 is Lake Pillans Wetland 1, and R5 is Lake Pillans Wetland. Quarry wetlands: Q1 is settlement dam, Q2 is quarry swamp, Q3 is tailings storage cell and Q4 is reservoir. Satellite image credit: Google Earth©.

perature, pH and conductivity (all log-transformed) as well as the sampling period as fixed effects. The addition of the sampling period removes inherent variability in the other factors (i.e. water temperature) between dates, providing more accurate estimates for the influence of the other factors on the response variable.

To analyse levels of DI, I used the methods outlined by Alford et al. (1999). I first partitioned measurement error and the variation representing the different kinds of asymmetry – directional asymmetry occurs when the mean of the left/right differences in sizes of bilateral structures is not ideally equal to zero (e.g. mammalian hearts; Graham et al., 1993); antisymmetry is in a structure that is less than perfectly symmetrical in the majority of individuals, but in any one individual, either side is equally likely to be larger (e.g. lobster claws; Palmer and Strobeck, 1986); and fluctuating asymmetry is found in structures that are ideally perfectly symmetrical, with left-right differences normally distributed with a mean of zero (i.e. a useful measure of DI). By calculating $F$-statistics from the appropriate mean square estimates, I determined (1) whether the degree of asymmetry can be used as an index of DI (i.e. if directional asymmetry is not present; $F = \mathrm{MS}_{\mathrm{SIDE}}/\mathrm{MS}_{\mathrm{SIDE}\times\mathrm{IND}}$). If $p < 0.05$, the degree of asymmetry cannot be used as an index of DI, as the ideal degree of symmetry cannot be known (i.e. structures appear to not be ideally bilateral). (2) Whether measurement error is small relative to developmental instability (fluctuating asymmetry and antisymmetry: $F = \mathrm{MS}_{\mathrm{SIDE}\times\mathrm{IND}}/\mathrm{MS}_{\mathrm{SIDE}\times\mathrm{IND}\times\mathrm{REP}}$). If $p < 0.05$, measurement error is small relative to the levels of fluctuating asymmetry and antisymmetry in the data, and thus, DI can be compared between populations. (3) How levels of DI differ between quarry and reference frog populations (quarry vs. reference: $F = \mathrm{MS}_{\mathrm{SAMPLE}\times\mathrm{IND}\times\mathrm{SIDE}}/\mathrm{MS}_{\mathrm{SAMPLE}\times\mathrm{IND}\times\mathrm{SIDE}\times\mathrm{REP}}$). If $p < 0.05$, levels of DI differ between populations. Here, SAMPLE is the quarry or reference wetland, SIDE is the left or right limbs, REP is replicate measurement number and IND is individual frog number.

Finally, I investigated the relationship between the abundance index for *C. signifera* and levels of DI using a general linear model with site as a blocking factor. Abundance index and DI were log-transformed. For all analyses, I assessed normality and homogeneity of variances using Q–Q and residual plots, respectively. All analyses were performed on R, Version 3.3.1 (R Development Core Team, 2016) using the lm function (Chambers, 1993). Data are available online (Sievers, 2017).

## 3 Results

### 3.1 Site characteristics

Quarry wetlands (mean ± SE; 6650 ± 2391 m) were on average approximately three times the size of reference (2200 ± 566 m) wetlands, whilst wetland perimeter was largely similar between quarry (394 ± 61 m) and reference (298 ± 67 m) wetlands. Although the precise age of several reference wetlands was unknown, all quarry wetlands were younger than reference wetlands (Table 1). During the two survey periods, water temperature was similar between quarry and reference wetlands, whereas pH and salinity varied considerably; Quarry wetlands were consistently more acidic and less saline than reference wetlands during both survey periods (Table 1). Vegetation cover within each spatial region (i.e. submerged, emergent, fringing and floating) was generally greater at reference wetlands.

### 3.2 Amphibian populations

Three amphibian species were calling and captured during the two survey periods: the common eastern froglet *Crinia signifera*, the southern brown tree frog *Litoria ewingii* and the striped marsh frog *Limnodynastes peronii*. All quarry wetlands contained at least one species, whereas only two of the five reference wetlands contained frogs (Fig. 2). Rarefaction curves quickly flattened out, suggesting that additional sampling would not considerably increase the number of species identified (Supplement Fig. S1). Multivariate analyses revealed little influence of the length of wetland perimeter or the proportion of emergent and fringing vegetation on the richness of amphibian communities, but quarry wetlands in general were significantly richer than reference wetlands (Table 2). Further, the pH of wetlands was a significant driver of species richness (Table 2). In terms of *C. signifera* abundance, wetland perimeter and the proportion of emergent and fringing vegetation were not significant predictors, but pH, treatment and the survey period were (Table 2), again with quarry wetlands harbouring more individuals than reference wetlands (Fig. 2).

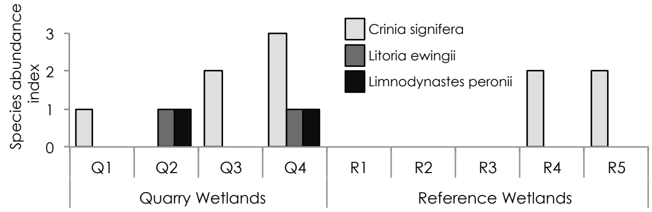


**Figure 2.** Abundance index for the common eastern froglet *Crinia signifera*, the eastern brown tree frog *Litoria ewingii* and the spotted marsh frog *Limnodynastes peronii* at quarry and reference wetlands. Abundance was estimated by categorising advertisement call intensity according to the index provided by Pope et al. (2000): (0) no individuals calling; (1) individual(s) can be counted with calls not overlapping; (2) calls of < 15 individuals can be distinguished, but there is some overlapping; and (3) > 15 individuals are calling.

### 3.3 Developmental instability

Only DI data for *Crinia signifera* were formally analysed (see Supllment Sect. S1 for all raw data). Analyses revealed no evidence of directional asymmetry in the frog population measured (forelimb: $p = 0.35$; hindlimb: $p = 0.58$; Table S1 in the Supplement), indicating that ideally, frog limbs should be symmetrical. Furthermore, measurement error was low relative to levels of fluctuating asymmetry and antisymmetry (forelimb: $p < 0.001$; hindlimb: $p < 0.001$; Table S1), indicating that a comparison between populations is permissible. Levels of DI were lower within quarry wetlands relative to reference wetlands (Fig. 3), with both forelimb ($p < 0.001$) and hindlimb asymmetry ($p = 0.027$) that are significantly different between populations (Table S1). Informal comparisons with values extracted from Alford et al. (2007) revealed that unsigned asymmetry in the quarry wetlands was similar to their control groups, whilst unsigned asymmetry in reference wetlands was similar to the impact group from Alford which suffered population decline following a period of increasing DI (Fig. 3). There was also a significant relationship between the abundance index for *C. signifera* and levels of DI, whereby indexed abundance was higher at sites with lower DI ($F_{1,11} = 17.9$, $p = 0.001$).

**Table 2.** Output from ANOVA for species richness and *Crinia signifera* abundance index with treatment (quarry or reference wetlands), wetland perimeter and vegetation (summed proportion of emergent and fringing vegetation) fitted as factors (Model 1). Treatment, water temperature, pH and conductivity, and date fitted as a blocking factor were included in Model 2. Asterisks indicate factors that were log-transformed to meet the assumptions of normality and homogeneity of variance. Boldface values represent significance at $\alpha = 0.05$.

| | Species richness | | | | Crinia signifera abundance index | | | |
|---|---|---|---|---|---|---|---|---|
| Factor | df | MS | *F* | *p* | df | MS | *F* | *p* |
| Model 1 | | | | | | | | |
| Treatment | 1 | 4.050 | 7.451 | **0.041** | 1 | 1.606 | 1.579 | 0.265 |
| Perimeter* | 1 | 0.007 | 0.013 | 0.915 | 1 | 0.864 | 0.850 | 0.399 |
| Vegetation | 1 | 1.226 | 2.255 | 0.194 | 1 | 0.001 | 0.001 | 0.981 |
| Residuals | 5 | 0.544 | | | 5 | 1.017 | | |
| Model 2 | | | | | | | | |
| Treatment | 1 | 7.004 | 19.834 | **0.001** | 1 | 12.604 | 24.828 | **0.001** |
| Temperature* | 1 | 0.000 | 0.001 | 0.976 | 1 | 0.096 | 0.189 | 0.673 |
| pH* | 1 | 3.168 | 8.970 | **0.013** | 1 | 3.423 | 6.742 | **0.027** |
| Conductivity* | 1 | 0.036 | 0.103 | 0.755 | 1 | 0.166 | 0.326 | 0.581 |
| Date | 1 | 0.698 | 1.975 | 0.190 | 1 | 3.073 | 6.053 | **0.034** |
| Residuals | 10 | 0.353 | | | 10 | 0.508 | | |

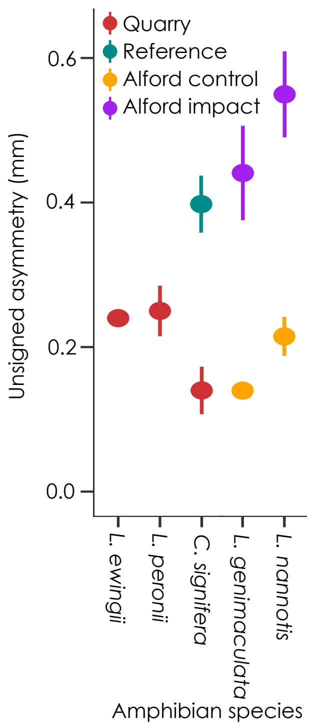


**Figure 3.** Levels of developmental instability (DI; quantified as unsigned asymmetry) for the three frog species occurring at quarry and reference wetlands. Mean DI values are also shown from Alford et al. (2007) for informal comparison. Frog populations from Alford control sites remained stable, whilst those from impact sites exhibited substantial decline following a period of increasing DI.

## 4 Discussion

Wetlands within the Kables Sands quarry were inhabited by more frog species than nearby urban wetlands, and the individuals within quarries had lower levels of developmental instability. Of the species present at the quarry, only *Crinia signifera* were present within reference wetlands, and these frogs had comparable rates of DI with frogs from populations that underwent crashes (Alford, et al., 2007). I also present evidence that there is a negative relationship between the abundance of *C. signifera* and unsigned asymmetry of their limbs at particular wetlands. Species occurrence and individual fitness proxies may be used to predict population persistence (Aldridge and Boyce, 2007). Therefore, quarry wetlands may provide more suitable habitat conditions than urban wetlands in the longer term.

Importantly, reference wetlands selected for this study do not necessarily represent pristine natural sites, but rather were artificial wetlands in suburban areas. Therefore, there may be additional pressures on these wetlands that quarry wetlands are not exposed to, such as elevated contaminants from storm water run-off, which influence amphibian richness, abundance and fitness (Hale et al., 2015). Despite not being pristine, amphibians frequently inhabit urban wetlands and these provide essential habitat in many urban areas (Brand and Snodgrass 2010; Le Viol et al., 2012). Since these reference wetlands are omnipresent in human-dominated landscapes, they serve as relevant and useful comparators to determine the suitability and ecological importance of quarry wetlands.

Typically, the mean abundance of *Crinia signifera* is greater at larger wetlands and at those with greater connectivity (Hamer et al., 2012). Although the site with the highest abundance of *Crinia signifera* – Q4 – was the second smallest quarry wetland by area, importantly, it also had the largest perimeter. Although there was no overall effect of wetland perimeter, given that this species calls from the waters edge, a greater perimeter may permit more individuals to be present and calling at any one time (Antis, 2013). Whilst salinity and pH were significantly higher within reference sites, and pH may have influenced frogs within this study, the levels observed within all surveyed wetlands were well within the range deemed suitable for the majority of amphibian species native to this region (Kearney et al., 2012; Sadinski and Dunson, 1992; Smith et al., 2007).

Our study and others conducted within Australian wetlands may shed light onto the factors driving community- and population-level patterns, and also provide information on how to best manage quarry wetlands. The occurrence of both *Limnodynastes peronii* and *Litoria ewingii* correlate with the proportion of vegetation in the emergent and submerged zones (Hamer et al., 2012). Although we found little influence of vegetation, at wetlands with $> 80\,\%$ cover, these species have a $> 0.8$ probability of being found, and yet were not heard calling at the reference wetlands that contained high levels of vegetation. In addition, *L. peronii* occurrence is correlated to the depth of wetland shores, where gently sloping shores are preferred over steep drops (Hamer et al. 2012). The shoreline of one of the reference wetlands, R4, was made up of a concrete wall, largely inappropriate for successful breeding by ground frog species (Parris, 2006). Within the quarry, there was considerable inter- and intra-wetland variability in the steepness of the shoreline, and given other species within the region (e.g. *Limnodynastes dumerlii*) prefer steep shores. This variability likely enhances year-round biodiversity in quarries. I provide considerable evidence that for sand quarry wetlands, multi-objective management may be highly beneficial for local amphibian populations (Bendell-Young, et al., 2000).

In addition to quantifying trends in species presence and population size within quarry wetlands, effective management and conservation also require knowledge of individual fitness. Studies have shown that DI increases as health decreases, and amphibian population declines have preceded periods of increasing DI (Alford, et al., 2007). I provide three lines of evidence suggesting quarry wetlands are providing a comparatively high-quality resource. Firstly, relative to reference wetlands, frogs within quarry wetlands had significantly lower levels of DI. Secondly, DI rates within quarry wetlands were comparable to the control populations from Alford et al. (2007). Frog populations within these control sites remained stable, whilst populations from impact sites exhibited substantial declines following a period of increasing DI. Finally, there was a significant relationship between levels of DI and the abundance of *C. signifera* populations. Although this suggests that population persistence and/or growth may be enhanced when DI is lower, quantifying population dynamics was beyond the scope of this study. Furthermore, whilst the incidence of amphibian abnormalities has been shown to be higher in wetlands impacted by other forms of mining (Anderson and Arruda, 2006; Rowe et al., 1996), I identified no abnormalities in any of the frogs captured in this study.

Empirical studies that monitor the relationship amongst population size, community composition, individual fitness and the environmental factors driving these responses are imperative to the long-term persistence of amphibians in human-impacted environments. Given that the quarry wetlands surveyed here provide a seemingly good habitat for amphibians, they – and similar wetlands elsewhere – should be managed to enhance their suitability and attractiveness. Gallagher et al. (2014) suggest ensuring that high-quality wetlands are connected to other wetlands, with their hydroperiod and vegetation managed to promote amphibian survival and reproductive success. Measures to remove and exclude exotic predators, such as mosquitofish and carp, will also greatly enhance amphibian populations and communities, with considerable improvements to individual fitness (Maezono and Miyashita, 2004; Tsunoda et al., 2010). To further promote and enhance amphibian biodiversity within quarries, it is important to provide substantial environmental heterogeneity; a mosaic of both wetland types and terrestrial refuges. Creating a diverse mosaic of wetlands that vary in size and depth will help ensure amphibian breeding at a proportion of quarry wetlands each season, thereby enhancing overall population and community persistence at any one quarry (Rannap et al., 2009). Despite little influence of vegetation on richness or abundance found here, creating wetlands with a minimum of 80 % emergent vegetation will help ensure a high probability of many native species occurring, such as *Limnodynastes peronii* and *Litoria ewingii* (Hamer et al., 2012). All species identified here oviposit their eggs within and around aquatic and semi-aquatic vegetation, which in turn provide refuge and protection to tadpoles (Anstis, 2013). Therefore, planting native vegetation around and within these wetlands would greatly increase the probability of colonisation. In quarries with existing wetlands, such as Kables Sands, implementing topographical modifications will ensure sufficient heterogeneous habitat to cater for as many species as possible.

In terms of monitoring, identifying amphibian species presence and population size and quantifying individual fitness by measuring limb lengths are economically and logistically feasible methods that can be used to assess the health of quarry wetlands. Measuring DI is a relatively simple process that could be employed periodically throughout the year at mining sites around the world. Coupled with the water quality data that are often already collected on site, levels of DI could provide an early warning indicator of potential issues, allowing mitigation strategies to be implemented prior to amphibian population declines. As researchers elsewhere adopt DI as a fitness proxy, these data can be used to compare levels observed in quarry wetlands, and ultimately, used to evaluate population health. Overall, the methods outlined here provide a powerful, yet simple, tool to assess the overall health of quarry wetlands that could be easily adopted and implemented at quarries anywhere in the world.

## 5 Conclusion

The creation of wetlands during and following mining activities is inevitable. Given the rate at which natural wetlands are being lost and degraded, the ecological importance of these wetlands has never been more important. Here, I show that quarry wetlands within the Kables Sands plant are not only attracting amphibians, but also providing conditions conducive to high individual fitness. Relative to other wetlands in this region, quarry wetlands consistently had greater species richness, as well as individuals with lower levels of developmental instability. Several management practices could further enhance the role quarry wetlands play in promoting amphibian biodiversity, such as planting out emergent zones with native vegetation, and creating a mosaic of wetland types. Implementing these basic survey techniques at quarries is an economically and logistically feasible strategy to ensure quarry wetlands are and remain high-quality systems capable of enhancing local and regional biodiversity and conservation goals.

**Data availability.** Data for DI analysis is in Sect. S1.

**The Supplement related to this article is available online at doi:10.5194/we-17-19-2017-supplement.**

**Competing interests.** The author declares that he has no conflict of interest.

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