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A conceptual framework for ecological responses to groundwater regime alteration (FERGRA)

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Abstract

Globally, the provision of groundwater-supported ecosystem services is threatened by climate change, water extraction, and other activities that alter groundwater regimes (defined as temporal dynamics in groundwater pressures, storage, and levels). Research on how altered groundwater regimes affect the ecology and ecosystem services of diverse groundwater-dependent ecosystems (GDEs) is currently fragmented with little integration across different GDEs, hampering our ability to understand and manage ecological responses to anthropogenic changes to groundwater regimes. To address this, we present a framework for assessing ecological responses to groundwater regime alteration (FERGRA). FERGRA is a logical approach to investigating how alterations to groundwater regimes change the timing, variability, duration, frequency, and magnitude of groundwater connections to different GDEs, in turn affecting their ecological processes and ecosystem service provision. Using FERGRA, multiple GDEs can be assessed concurrently, optimizing their integrated management. Unifying the concepts of ecological responses to altered groundwater regimes and groundwater connections of different GDEs across the landscape, FERGRA provides a framework for (a) organizing the currently fragmented research on GDEs to better identify commonalities and knowledge gaps, (b) formulating and testing hypotheses for quantifying ecological responses to groundwater regime alteration in GDEs to derive general principles to guide research and management, and (c) facilitating assessments of the trade-offs between the benefits of groundwater extraction (e.g., to support mining and agriculture) versus conservation of GDEs to protect other ecosystem services.

KEYWORDS

aquifers, fish, groundwater depletion, groundwater-dependent ecosystems, macroinvertebrates, phreatophytes, streams, stygofauna

1 | INTRODUCTION

Groundwater-dependent ecosystems (GDEs) are ecological communities and associated biophysical processes that require permanent or intermittent connection to groundwater to persist. These ubiquitous ecosystems are ecologically diverse (Figure 1), ranging from groundwater-dependent vegetation (Eamus, Zolfaghar, Villalobos-Vega, Cleverly, & Huete, 2015), coastal ecosystems receiving submarine groundwater discharge (Basterretxea et al., 2010), and wetlands, springs, and baseflow-fed rivers (Boulton & Hancock, 2006) through to confined and unconfined aquifers containing unique groundwater biota (stygofauna; Korbel & Hose, 2015). GDEs can be classified into (a) ecosystems reliant on the surface expression of groundwater (e.g., rivers, wetlands, springs, and some estuarine and coastal ecosystems), (b) ecosystems reliant on the subsurface presence of groundwater (e.g., floodplain, riparian, and terrestrial vegetation),

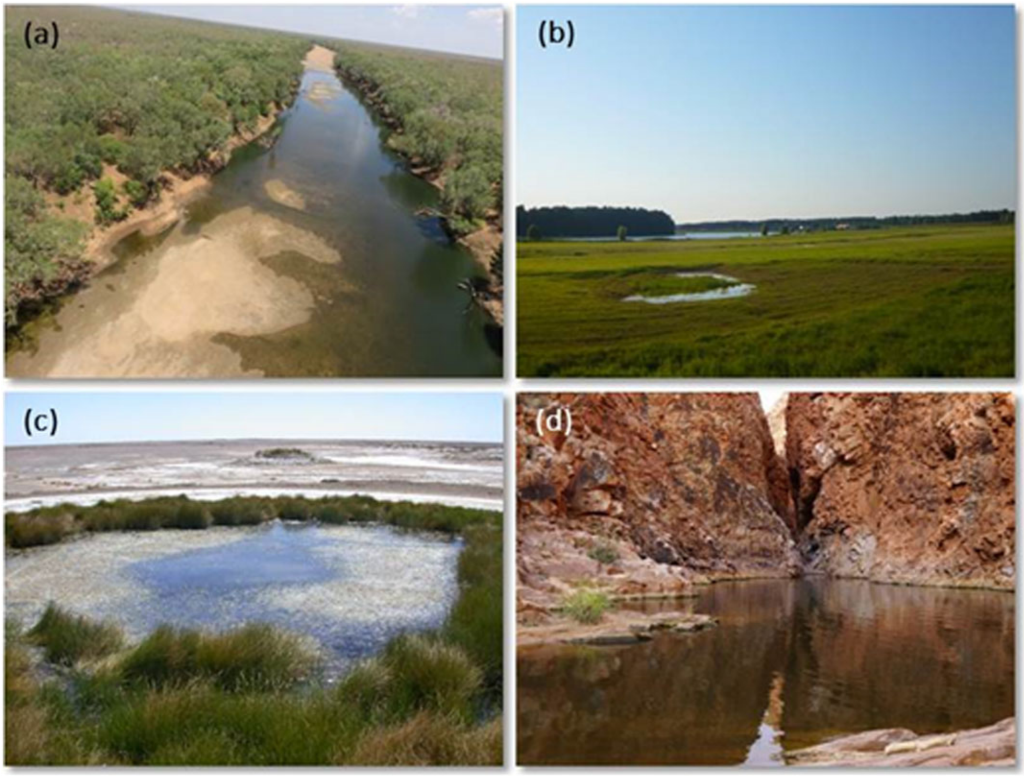

FIGURE 1 Groundwater connections support a wide diversity of ecosystems, including (a) river sections that would otherwise be dry (Fitzroy River, north-western Australia), (b) riparian and floodplain vegetation (Elbe River, northern Germany), (c) mound springs (South Australia), and (d) arid-zone waterholes (central Australia)

and (c) aquifer and cave ecosystems partially or completely inundated with groundwater (Eamus, Froend, Loomes, Hose, & Murray, 2006). This classification spans the spectrum of GDEs and their varying hydrological connectivity to groundwater.

Hydrological connectivity to groundwater underpins GDE functioning and governs fundamental ecological processes such as nutrient cycling, water storage, and provision of habitat for diverse biota (Boulton et al., 2014; Mclaughlin & Zavaleta, 2012). In turn, these ecological processes and biodiversity support the many ecosystem services provided by GDEs, ranging from water purification and provision of food and fibre through to climate regulation and cultural services such as tourism and recreation (Griebler & Avramov, 2015). Provision of these services relies on hydrological connectivity between groundwater and GDEs, mediated by what we term the "groundwater regime." Groundwater regimes encompass the temporal variability in magnitude, frequency, duration, timing, and rate of change of groundwater levels, storage, and pressures at given spatial scales (analogous to streamflow regimes,e.g., Poff et al., 1997). We posit that these regimes control hydrological connectivity between groundwater and the three broad classes of GDEs described above.

Human activities, especially groundwater extraction, alter natural groundwater regimes. Extraction has drastically changed groundwater regimes in China, India, Pakistan, most countries in North Africa and the Middle East, and much of the Americas, Africa, and Australia (Gleeson, Wada, Bierkens, & van Beek, 2012). In the Ganges–Brahmaputra Basin alone, groundwater extraction reduces aquifer storage by around 12 km^3 annually (Richey et al., 2015). Burgeoning rates of global groundwater extraction have sparked demands to predict ecological responses to groundwater regime alteration worldwide (e.g., Alley, Healy, LaBaugh, & Reilly, 2002; Gleeson et al., 2012). In particular, there is a need for better understanding of relationships between groundwater regimes and GDEs so that management guidelines, such as groundwater extraction thresholds, can be derived (Brown, Bach, Aldous, Wyers, & DeGagné, 2010). Although some of these relationships have been derived for aspects of individual GDEs (e.g., groundwater-dependent fens in central Oregon; Aldous & Bach, 2014), we lack a collective framework to organize our understanding of the consequences of groundwater regime alteration on different GDE classes co-occurring in the landscape. This contrasts with other relevant fields that use frameworks to synthesize information on responses to disturbance and guide hypothesis development and management. Examples include ecological responses to streamflow regime alteration (Poff et al., 2010) and forest responses to altered fire regimes (McWethy et al., 2013).

To address this, we present a framework for assessing ecological responses to groundwater regime alteration (FERGRA), and use it to generate hypotheses about ecohydrological mechanisms by which disruptions of GDE–groundwater connectivity affect the provision and transfer of ecosystem services from GDEs across the landscape. Information collected to test these hypotheses is then used to guide landscape-scale (e.g., across a catchment or multiple conjoined catchments) management of GDEs and inform decision-makers about trade-offs between the benefits of groundwater extraction versus conservation of GDEs to protect other ecosystem services. After outlining six scientific premises underpinning the framework, we describe the five steps in applying FERGRA, illustrated with worked examples. We conclude by reviewing the framework's applications where multiple GDEs co-occur in the landscape.

2 | SCIENTIFIC PREMISES UNDERPINNING FERGRA

FERGRA is underpinned by six scientific premises (Table 1) that provide the logical basis of the ecohydrological mechanisms underlying the framework (Figure 2). The first premise is that multiple, rather than single, interacting components of the groundwater regime influence biodiversity and ecological processes in GDEs, operating primarily when groundwater is hydrologically connected to the GDEs. The second is that GDE–groundwater connectivity (i.e., when a GDE is in hydrological contact with groundwater) ranges from continuous (e.g., permanently saturated aquifers; Humphreys, 2006) to intermittent (e.g., rivers seasonally connected to groundwater; King, Townsend, Douglas, & Kennard, 2015) and is governed by the groundwater regime. Premise 3 is that alteration of the natural variability in the groundwater regime and groundwater–GDE hydrological connectivity typically reduces biodiversity and impairs ecosystem functioning in GDEs (Figure 2), paralleling the situation observed when the natural variability in streamflow regimes is altered (King et al., 2015; Poff et al., 2010). The fourth premise is that these reductions in biodiversity and impairment of ecosystem functioning in GDEs reduce provision and/or the flows of many ecosystem services as portrayed in the “cascade model” of Haines-Young and Potschin (2010).

It is essential for managers to quantify groundwater regime alterations and groundwater–GDE connectivity, together with the ecological impacts of these alterations; otherwise, actions to conserve GDEs risk being misdirected and unable to demonstrate benefits. Therefore, Premise 5 is that these changes are quantifiable using various techniques (e.g., Eamus et al., 2015) combined with biological indicators such as biodiversity and rates of ecosystem processes to measure ecological responses (Figure 2). The final premise is that assessing ecological responses to groundwater regime alteration in multiple GDEs co-occurring in the landscape is essential for successful integrated management of these ecosystems and sustaining their ecosystem services (Table 1). All six premises are linked (Figure 2), providing a logical basis for FERGRA and its application to ecosystem management of different GDE classes.

3 | THE FIVE STEPS OF FERGRA

There are five steps in the framework (Figure 3), a logical sequence that describes the groundwater regime, how it has been altered, how this alteration currently or potentially affects groundwater connectivity to one or more GDEs, and what ecological responses might result. This information provides the basis for deriving hypotheses and targeting data collection to test them and leads to the final step of making management recommendations, including assessment of trade-offs in decision making (Figure 3). The process is iterative, with information shared among the steps.

The logic resembles that currently being applied by The Nature Conservancy in Oregon and the U.S. Forest Service to develop and test approaches for determining groundwater flows and levels required to sustain freshwater ecosystems (Aldous & Bach, 2013), which follows four steps: (1) ascertain the amount and timing of groundwater flow required to support GDE species and ecosystem processes, (2) assess how groundwater flow into the GDEs will be altered under different withdrawal scenarios, (3) determine the amount of ecological change expected at different levels of withdrawal, and (4) decide on the acceptable level of change in ecological conditions and ecosystem function. FERGRA extends this to multiple different GDEs concurrently to enable the landscape-scale analysis appropriate for regional management and uses a hypothesis-based approach to assess ecological responses to changes in hydrological connectivity between GDEs and groundwater caused by altered groundwater regimes. Although FERGRA shares similar management goals, it is intended to be more collective and span multiple types of GDEs concurrently in the same landscape and could also incorporate the effects of altered groundwater quality (see later). As the focus of FERGRA is on altered groundwater regimes rather than solely groundwater withdrawal, it can also be applied where water tables are artificially raised (e.g., aquifer reinjection) or groundwater pressures are changed (e.g., coal-seam gas extraction).

Application of the framework is illustrated with a worked example using publicly available groundwater data (Queensland Government, 2017) from the Condamine catchment, Australia. This catchment has undergone extensive groundwater regime alterations with the expansion of irrigated agriculture since the 1960s (Dafny & Silburn, 2014; Le Brocque, Kath, & Reardon-Smith, 2018), exemplifying a common type of groundwater regime change that is occurring globally and thus is

TABLE 1 Six scientific premises underpinning the ecohydrological mechanisms underlying FERGRA

Premise
1: Biodiversity and ecological processes in GDEs are influenced by multiple interacting components of the groundwater regime, primarily when groundwater is hydrologically connected to the GDEs.
2: GDE–groundwater connectivity ranges from continuous (e.g., aquifers) to intermittent (e.g., groundwater-dependent vegetation) and is governed by the groundwater regime.
3: Alteration of the natural variability in the groundwater regime (and GDE–groundwater connectivity) typically reduces biodiversity and impairs ecosystem functioning in GDEs.
4: Reductions in biodiversity and impairment of ecosystem functioning in GDEs reduces or alters provision and transfers of many ecosystem services.
5: Various techniques (e.g., hydrogeological monitoring, geochemical tracers, and modelling) can be used to quantify alterations of groundwater regime and GDE–groundwater connectivity, the ecological impacts of which can be measured using various indicators (e.g., biodiversity and rates of ecosystem processes).
6: Assessing ecological responses to groundwater regime alteration in multiple GDEs co-occurring in the landscape is essential for integrated management of these ecosystems and sustaining their ecosystem services.

Note. Relationships among premises are shown in Figure 2. GDE: groundwater-dependent ecosystem.

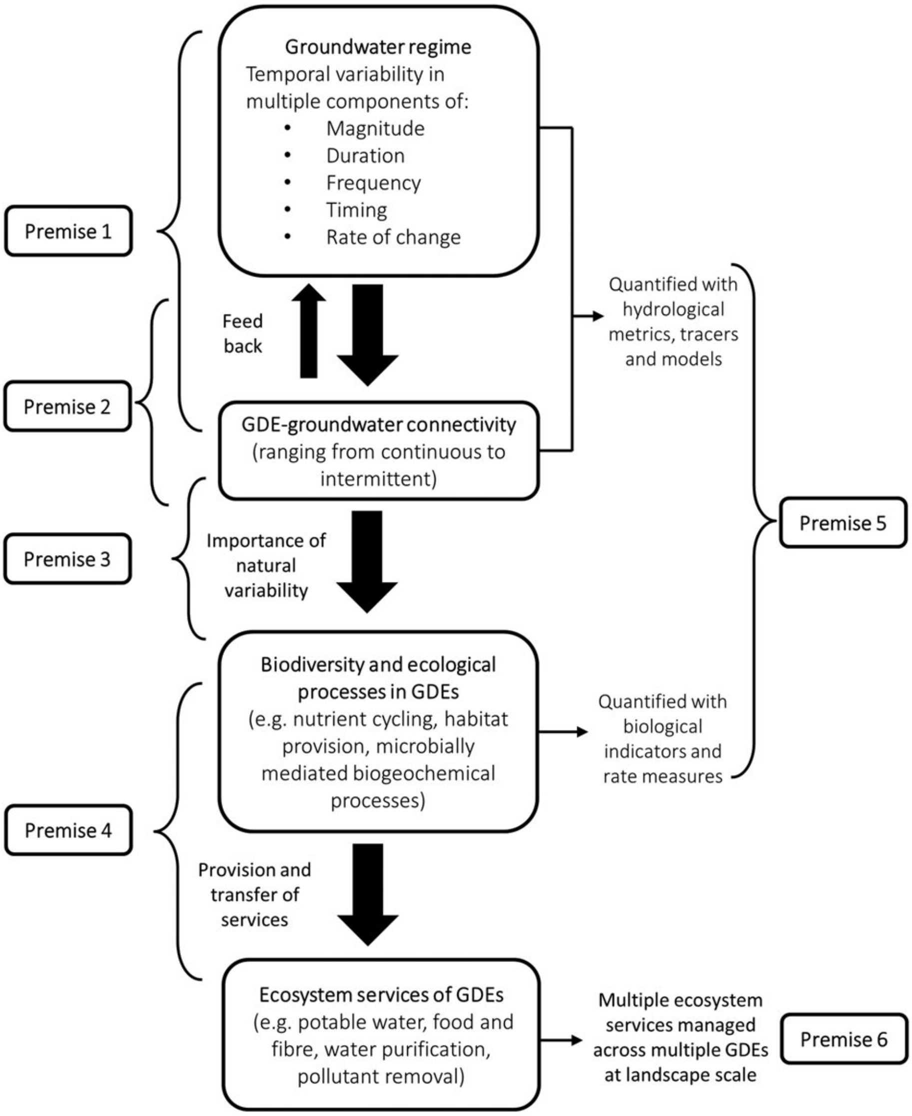

FIGURE 2 The six premises that underlie the framework for ecological responses to groundwater regime alteration (FERGRA). The width of the broad arrows represents their hypothesized relevance

useful for illustrating the concepts of the groundwater regime and the five steps of FERGRA. The example uses groundwater data observations from a long-term (1970 to 2014) monitoring bore in the area. As linked ecological and groundwater data for GDEs, aside from vegetation, are lacking, we have extended the vegetation data with hypothetical examples of GDE–groundwater connections and ecological GDE responses to demonstrate how groundwater regime and ecological data may be analysed using FERGRA across multiple different GDE classes in a landscape.

3.1 | Step 1: Hydrogeological foundations: Describing the groundwater regime

The first step involves collating and assessing knowledge of the groundwater regime relevant to the GDEs of interest. Analogous to the streamflow regime (Poff et al., 1997), the groundwater regime describes temporal dynamics in timing, frequency, magnitude, variability, rates of change, and durations of minima, means, and maxima for

a. water table depths (e.g., distance between the water table [upper surface of the saturated zone] and GDEs [e.g., their average or species-specific root depths]);
b. groundwater volumes (e.g., volumes of the saturated zone); and/or
c. groundwater hydraulic heads and flow rates (e.g., groundwater dynamics and fluxes).

Water table depth regimes vary widely across different topographical, climatic, and hydrogeological contexts (e.g., Fan, Miguez-Macho, Weaver, Walko, & Robock, 2007; Taylor et al., 2013). Groundwater extraction also causes major long-lasting changes to water table depths and dynamics. For example, groundwater extraction in India and Bangladesh has increased water table depths by 0.1–0.5 m yr^{-1} (Rodell, Velicogna, & Famiglietti, 2009). Some scenarios of groundwater extraction for the High Plains aquifer (U.S.) predict reductions of 100–200 m in aquifer saturated thickness by 2110 (Steward et al., 2013). Altered dynamics and lowered water tables have been associated with changes in GDE vegetation composition, condition

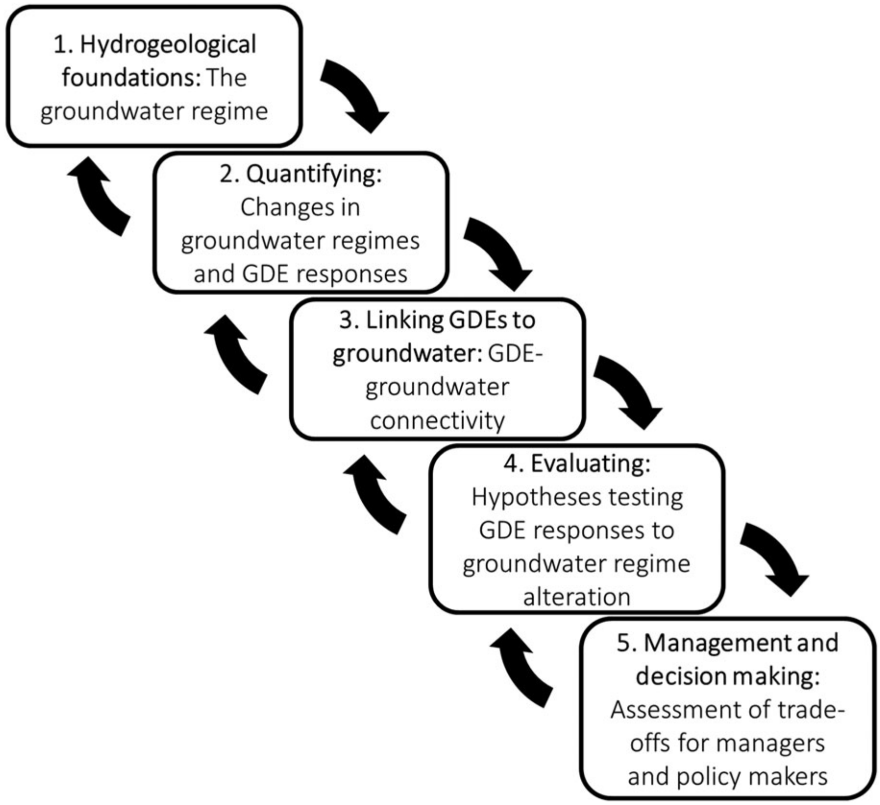

FIGURE 3 Overview of the five steps of FERGRA. Arrows show flow of information between steps and the iterative nature of the framework

and function (Eamus et al., 2015), and impaired stream condition (Falke et al., 2011).

Aquifer storage varies by orders of magnitude depending on geology, void characteristics, recharge and discharge rates, and catchment land use (Alley et al., 2002; Boulton et al., 2014). Again, extraction has substantially reduced groundwater storage in many regions (Steward et al., 2013). Global groundwater depletion from 2000 to 2009 is estimated at 113 $km^3\ yr^{-1}$, with highest rates of change in India, the United States, Iran, Saudi Arabia, and China (Döll, Müller Schmied, Schuh, Portmann, & Eicker, 2014). Reduced storage affects GDEs by lowering rates of groundwater discharge into GDEs such as rivers and wetlands (Boulton & Hancock, 2006) and reducing the amount of saturated habitat for obligate groundwater-dependent biota (Korbel & Hose, 2011, 2015).

Groundwater hydraulic head is a measure of the potential energy in groundwater flow systems and is the combination of the gravitational and pressure potentials (Hiscock & Bense, 2014). Pressure potentials can also be influenced by density differences caused by variations in salinity and temperature. The hydraulic head at a given point is approximated by the water level in a well or piezometer. In unconfined aquifers, variations in the hydraulic head can be used to represent changes in the water table. Spatial and temporal variations in groundwater hydraulic heads and fluxes largely control GDE–groundwater connectivity. For example, where the water table intersects a stream channel, differences in hydraulic head either increase or decrease recharge to the groundwater from the stream (Alley et al., 2002), with major repercussions for biota and ecological processes in the river and its hyporheic zone (Larned, Gooseff, Packman, Rugel, & Wondzell, 2015).

Variations across these three components of the groundwater regime (water table depths, groundwater volumes, and groundwater hydraulic heads and flow rates) will not always correspond. For example, GDEs connected to the same aquifer affected by the same drawdown level (change in groundwater depth) may experience different changes in groundwater flow rates depending on the conductivity of the material between the source aquifer and each GDE.

Temporal dynamics in these three components can be visualized using what we term a "hydrogeograph" (Figure 4). We introduce this new term, applying it in the context of groundwater and GDEs in the same way that hydrologists use streamflow hydrographs to show changes in discharge or river level over time (Gordon, McMahon, &

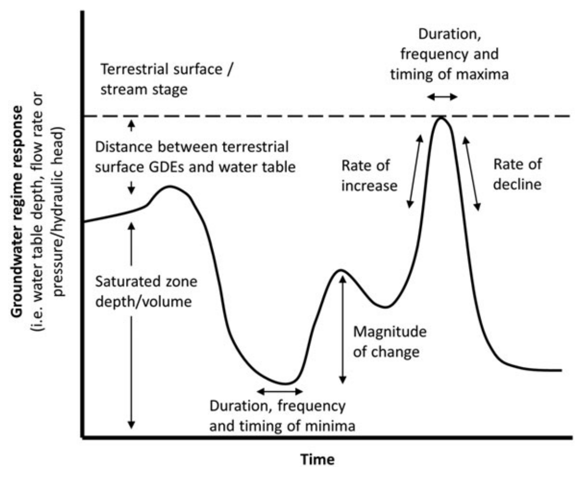

FIGURE 4 Hydrogeograph showing the components of the groundwater regime

Finlayson, 2004). Trends portrayed on a hydrogeograph can illustrate groundwater–GDE connectivity for multiple GDEs in a landscape subject to groundwater regime alterations (Figures 4 and 5) in the same way hydrographs have been used to describe surface-water connectivity in riverine ecosystems (e.g., Ward & Stanford, 1995).

Groundwater regime variables must also be considered within their hydrogeological and landscape contexts. Recharge and discharge properties, as well as fundamental hydrogeological properties, can all influence groundwater regime dynamics (Crosbie, Davies, Harrington, & Lamontagne, 2015). Furthermore, groundwater recharge and water table dynamics can vary depending on soil properties (e.g., percentage of clay) and landscape properties (e.g., native vegetation, cropping, or pasture) and be more variable in recharge areas compared with discharge areas (Crosbie et al., 2015). Thus, empirical groundwater metrics by themselves may not adequately reflect the groundwater regime; landscape and hydrogeological properties also need to be considered when assessing and measuring changes in groundwater regimes.

Our framework focuses on hydrological attributes because this parallels conventional assessments of surface-water regimes described by water budgets, hydrographs, and most hydrogeological models, allowing FERGRA to be integrated with surface-water concepts and models. However, FERGRA can also include assessment of water quality attributes (e.g., salinity, pH, and contaminant concentrations) and be applied to polluted groundwater. GDEs' ecological relationships with groundwater regimes can differ depending on groundwater quality (Menció, Korbel, & Hose, 2014). For example, GDEs that respond positively when connected with unpolluted groundwater may be negatively impacted if connected to polluted groundwater (Kath et al., 2015).

Interactions between groundwater pollution and hydrological groundwater regime alteration often have important consequences for GDEs. Secondary salinization can increase groundwater inflows, and thus, the groundwater table and inundation may increase alongside declines in water quality (Jolly, McEwan, & Holland, 2008). FERGRA could be applied in areas of groundwater pollution to understand the ecological benefits of minimizing connections between vulnerable GDEs and groundwater. The consequences of interactions between groundwater regime alteration and pollution is an important topic for future research, but from here, we focus on the hydrological aspects of groundwater regime alteration to be consistent with other frameworks focused on ecological responses (e.g., ELOHA; Poff et al., 2010).

3.1.1 | Case study Step 1: Groundwater regime data from the Condamine catchment

Figure 6 shows an example of a hydrogeograph plotted from 45 years of groundwater depth observations from our Condamine catchment case study. The groundwater regime data were aggregated to monthly values, grouped into 5-year classes (dashed lines on Figure 6). As in many other parts of the world, the key drivers of groundwater regime alteration in the Condamine are linked to water extraction to support irrigated agriculture (Döll et al., 2014).

The hydrogeograph shows general characteristics of the groundwater regime (cf. Figures 4 and 5), illustrating periods of persistent groundwater decline (ca. 1980–2005) interspersed with periods of relatively rapid increases in groundwater depths (Figure 6). These trends in groundwater regime are likely linked to changes in climate and land use that have influenced the rates of groundwater recharge and groundwater use (Dafny & Silburn, 2014; Le Brocque et al., 2018). Further groundwater regime data (not shown) were taken from two other bores (Queensland Government, 2017) to be linked with remotely sensed riparian data as described later (Case study Step 2).

3.2 | Step 2: Quantifying changes in the groundwater regime and ecological responses of GDEs

Monitoring data and hydrogeological models can be used to quantify the changes in the groundwater regime shown in Figures 4–6. Hydrogeological monitoring data that quantify declines in

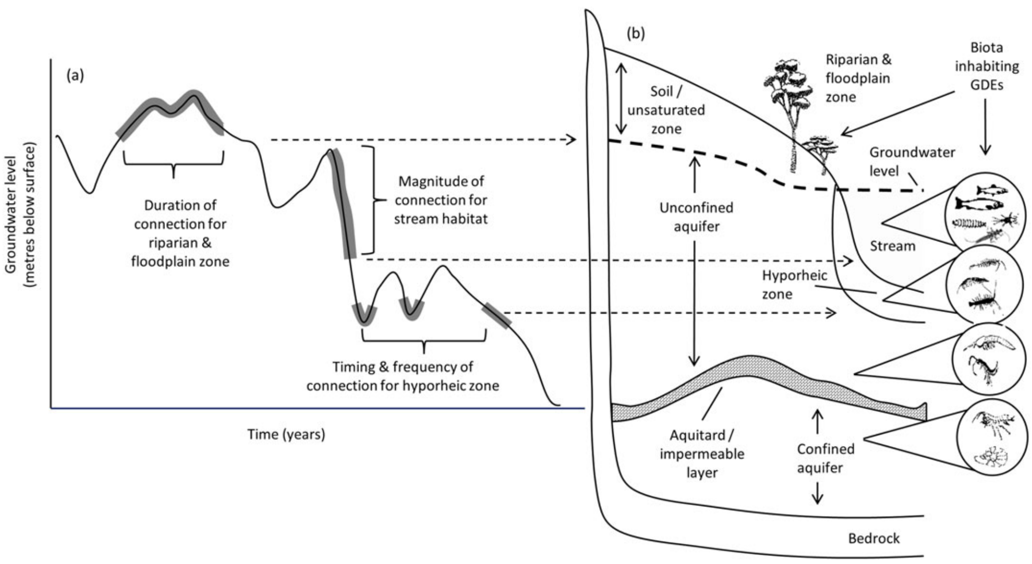

FIGURE 5 (a) Hydrogeograph showing changes in groundwater level, which may have resulted from natural and/or anthropogenic changes through time for an unconfined aquifer. Grey areas show associations between groundwater regime components and possible links (dashed arrows) with different GDEs based on hypothetical levels of GDE–groundwater connectivity. In (b), a cross section of a landscape shows how the hydrogeograph is associated with different GDEs and their biota across a landscape

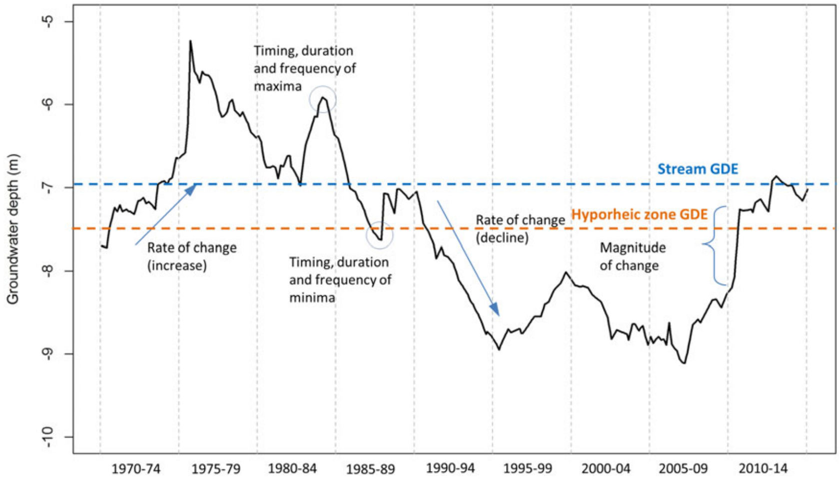

FIGURE 6 Hydrogeograph for the case study example. The black solid line is monthly groundwater depth for the Condamine bore (#42230860; Queensland Government, 2017). Vertical grey dashed lines delineate each of the 5-yearly segments for which groundwater regime metrics have been calculated. Horizontal dashed lines represent hypothetical GDE-connectivity levels for the stream GDE (blue line) and the hyporheic zone GDE (orange line); the GDE-connectivity level for the aquifer GDE (−30 m) is not shown

groundwater pressure triggered by groundwater extraction (Bennett & Gardner, 2015) can be used to build hydrogeological models (Tian et al., 2015) to predict how climate, land use, and/or extraction drive alterations to groundwater regimes represented by changes in water table depths (e.g., Fan et al., 2007), aquifer storage (e.g., Richey et al., 2015), and groundwater pressures (e.g., Alley et al., 2002). For example, the influences of land use changes and water extraction practices on groundwater storage in Rajasthan, Punjab, and Haryana were modelled using data from NASA Gravity Recovery and Climate Experiment satellites and simulated soil-water variations from a data-integrating hydrological model (Rodell et al., 2009).

The linking of monitoring data and models of groundwater regime dynamics to measurements of GDE ecological responses is in its infancy, although the importance of linking hydrogeological models to data on ecological responses across multiple GDEs has been recognized (e.g., Tian et al., 2015). Space-for-time approaches, where spatial variations in depth to groundwater are used as a surrogate for what might happen if groundwater depths changed over time, have been used to infer GDE responses to longer-term groundwater changes (e.g., Eamus et al., 2015). Matched sets of hydrogeological and ecological data are becoming more widely available, facilitated by technologies such as remote sensing for efficiently gathering these data at multiple scales (Crosbie et al., 2015; Seidl, 2017).

However, assessments of groundwater regime dynamics and groundwater connections to multiple different GDEs in the same landscape as FERGRA proposes are rare in the literature. Instead, most studies focus on single attributes of the groundwater regime and single GDEs (or one class of GDEs), usually water table depth and groundwater-dependent vegetation (e.g., fens [Aldous & Bach, 2014] and alkali meadows [Elmore, Manning, Mustard, & Craine, 2006]). Modelling is often used and can combine historical and predicted data on groundwater regime to forecast changes in components of GDEs (e.g., effects of groundwater pumping from the U.S. High Plains aquifer on Great Plains stream fish assemblages; Perkin et al., 2017). In another example, Chan et al. (2012) used a groundwater and surface-water hydrodynamic model that incorporated monitoring data to assess the impacts of water extraction on stream fish responses.

One key challenge to overcome when linking ecological responses to groundwater regime dynamics is the frequent mismatches in the scale and resolution of available groundwater and ecological data. Measurements of aquifer storage are usually made at broad scales (e.g., spatial resolutions of 0.5° × 0.5° [55 km × 55 km at the equator]; Döll et al., 2014), but these are difficult to reconcile with fine-scale ecological responses such as changes in abundance, condition or composition of groundwater-dependent vegetation (Eamus et al., 2015), aquatic fauna in springs (Powell, Silcock, & Fensham, 2015), or instream and wetland fauna that are all typically assessed at scales <1 km^2. In some cases, ecological data can be aggregated to larger scales (e.g., reaches or catchments) and linked with broad-scale groundwater data, but this might not always be commensurate with scales needed for management and often entails loss of important information on fine-scale spatial variability. Overcoming the problem of mismatches in the scales at which groundwater and ecological data are available requires concurrent data collection at the relevant scales. FERGRA provides a systematic framework to identify scale mismatches explicitly, guiding the subsequent collection of matched data at broader or finer scales as needed.

A second challenge is that ecological responses to groundwater regime alteration are often nonlinear (e.g., Kath et al., 2014) and accelerate sharply after a particular threshold is exceeded (Aldous & Bach, 2014). Detection of these thresholds is sometimes difficult because there may be lag effects where decades elapse before ecological impacts of groundwater regime alterations are evident (Alley et al., 2002). Again, spatial scale is relevant because the various components (e.g., table 2 in Korbel & Hose, 2011) of different GDEs are likely to respond at different scales. Ascertaining GDE components that are particularly sensitive to alterations of groundwater regime is an important goal when selecting potential indicators, especially those that may serve as "early warning" in vulnerable GDEs and can be used to trigger management responses.

Potential ecological indicators of groundwater regime alteration exist for all three broad classes of GDEs. The best-studied indicators are abundance and/or condition of various groundwater-dependent vegetation species and communities in response to altered groundwater regimes (e.g., Aldous & Bach, 2014; Barbeta et al., 2015; Brown et al., 2010; Eamus et al., 2015; Elmore et al., 2006). Less well studied are the potential ecological indicators in riverine and wetland GDEs; these include densities of particular groups of fish (e.g., Falke et al., 2011; Perkin et al., 2017) and macroinvertebrates (e.g., Stubbington, Wood, Reid, & Gunn, 2011) as well as measures of groundwater-influenced ecosystem processes such as organic matter decomposition in the hyporheic zone (e.g., Burrows et al., 2017). Densities and community composition of stygofauna are commonly proposed as ecological indicators of altered groundwater regimes in cave (e.g., Chilcott, 2013) and aquifer GDEs (e.g., Korbel & Hose, 2015; Tomlinson, Boulton, Hancock, & Cook, 2007), although there are other potential indicators such as microbial activity (Korbel & Hose, 2011; Lategan, Korbel, & Hose, 2010).

The rapidly developing interest in using traits to understand ecological responses to disturbances (Schmera, Podani, Heino, Erős, & Poff, 2015) could be extended to GDEs. In more hydrologically dynamic GDEs, species with traits such as sensitivity to desiccation or poor dispersal ability may be vulnerable to groundwater regime alteration and therefore make especially effective ecological indicators (Brunke, Hoehn, & Gonser, 2003; de Szoeke, Crisman, & Thurman, 2016; Stumpp & Hose, 2013). Poor dispersers may lack the mobility to track favourable hydrological conditions and be restricted to hydrologically stable streams with groundwater regimes that buffer against precipitation variability (Kath et al., 2016). Other potential indicators in GDEs are the rates of ecological processes influenced by groundwater regime alteration. Such processes include organic matter decomposition and nutrient cycling (Griebler & Avramov, 2015) and often underpin provision of GDE ecosystem services (Figure 2) that would be impaired by alteration of the natural groundwater regime.

3.2.1 | Case study Step 2: Quantifying groundwater regime alteration and ecological responses in multiple GDEs in the Condamine catchment

Once groundwater regime data have been collated (Case study Step 1), the next step is to quantify concurrent hydrogeological and ecological changes. We used R (R Development Core Team, 2016) to quantify changes in different groundwater regime metrics (see Appendix S1 for codes for quantifying groundwater regime components). In this example, the groundwater regime data were divided into 5-year segments (Figure 6) to calculate groundwater regime metrics. Grouping of the data is arbitrary and could be based on shorter or longer time periods (or at different locations). Rates of change in groundwater level were the most negative from 1990 to 1994 at −0.03 m per month in 1970–1974 and most positive at 0.02 m per month in 2010–2014 (Figure 6). The greatest magnitude of change occurred in the 1990–1994 period when the groundwater level fell by −1.74 m (Figure 6). Long durations of maxima are notable in 1980–1984 and 2010–2014, whereas long durations of minima occurred in 1985–1989 and 1995–1999 (Figure 6).

As actual ecological data relating to GDEs in our Condamine example are limited, we have supplemented them with hypothetical conceptual examples to reflect the types of ecological responses that have been measured in other studies and could serve as additional indicators of groundwater regime alteration (cf. table 2 in Korbel & Hose, 2011). To span the full range of GDE classes and to include process measurements as well as biotic data, our conceptual examples simulated (a) a stream GDE with quantitative data on aquatic macroinvertebrates from multiple reaches of the Condamine River sampled in different seasons, (b) a hyporheic zone GDE in an intermittent stream with data on organic matter breakdown rates measured as leaf litter breakdown and cotton-strip decomposition, and (c) an aquifer GDE with stygofauna species-richness data from repeated net hauls and pump samples in eight bores (up 36.6 m deep) sampled seasonally. These conceptual examples draw on published literature either directly from the Condamine River system (stream macroinvertebrates—Chessman, Jones, Searle, Growns, & Pearson, 2010; Marshall, Steward, & Harch, 2006) or GDEs in eastern Australia whose biota and ecosystem processes are likely to resemble those in the Condamine catchment (hyporheic GDE—Burrows et al., 2017; aquifer GDE—Hancock & Boulton, 2009). Detailed sampling methods for quantifying the dependent variable specified for each conceptual example of GDE class are described in these four references, along with sufficient raw data to infer plausible outcomes for the next three steps (see later) for these three GDE classes in the Condamine catchment case study.

The GDE vegetation case study example uses remotely sensed enhanced vegetation index (EVI) data from riparian forests in the Condamine catchment. As EVI data show changes in green biomass, they can be used to monitor condition in vegetation GDEs (e.g., Elmore et al., 2006). These data were derived from the MODIS Terra sensor using the LPDAAC MOD13Q1 product (Lymburner et al., 2010) at a nominal 250-m resolution. The MOD13Q1 (16 days maximum value composite EVI) was stacked date sequentially for the period 2000–2011, and the stack was filtered for spikes and drops in the data time series. We used data from seven sites (each with ~340 observations) of mapped *Eucalyptus camaldulensis*, a widespread riparian tree commonly dependent on groundwater (Kath et al., 2014). All seven sites were within 1.5 km of two monitoring bores (bore numbers #42230148 and #42231210; Queensland Government, 2017) from which we derived groundwater regime data.

3.3 | Step 3: Linking the groundwater regime to GDE responses: Determining GDE–groundwater connectivity

The third step in FERGRA is to determine GDE–groundwater connectivity to assess the hydrogeological association of altered groundwater regimes with ecological responses by multiple GDEs in the landscape. The importance of groundwater connectivity for the persistence and functioning of the different classes of GDEs (Table 1, Figure 2) is demonstrated by changes in the habitat and condition of riparian vegetation (Eamus et al., 2015), fish (Falke et al., 2011), macroinvertebrate (Brunke et al., 2003), and stygofauna (Stumpp & Hose, 2013) communities when groundwater connectivity is altered. Within FERGRA, we hypothesize that alterations to GDE–groundwater connectivity are the primary ecohydrological mechanism by which groundwater regime alterations elicit ecological responses in all classes of GDEs.

3.3.1 | Components of GDE–groundwater connectivity

We propose four main components of GDE–groundwater connectivity. The first is the timing when a GDE is connected to groundwater. This is likely to be important in systems with strong seasonal and other temporal ecological cues (e.g., fish reproduction and migration; King et al., 2015). Timing of groundwater connections may also strongly influence the biota of GDEs subject to seasonal or supraseasonal precipitation droughts, especially when groundwater connections maintain drought refuges in GDEs (Barbeta et al., 2015) or favour particular taxa (e.g., the amphipod *Gammarus pulex* in a groundwater-fed English stream; Stubbington et al., 2011).

Second is the frequency of GDE–groundwater connections, defined here as how often a specified GDE or suite of GDEs is connected to groundwater over a given time period. GDEs may need a minimum number of groundwater connections annually or seasonally to ensure persistence such as a minimum frequency of connections to groundwater by GDE vegetation to maintain physiological functioning during establishment (e.g., Segelquist, Scott, & Auble, 1993).

The third component is the duration of unbroken GDE–groundwater connections, which could determine the ecological integrity and persistence of GDEs. Periods of sustained groundwater connection may be needed to maintain flow permanence in GDE-dependent stream sections that support instream biota (Falke et al., 2011; Stubbington et al., 2011) or biogeochemically active hyporheic zones (Burrows et al., 2017). Changes in the duration of groundwater connections may also alter groundwater-dependent vegetation composition, with implications for its habitat potential (Merritt & Bateman, 2012).

The fourth component is the magnitude and rate of change in attributes of the groundwater regime during GDE–groundwater connection. This component is intended to capture the importance of groundwater regime dynamism to different classes of GDEs. Alterations in the magnitude and rate of change can influence the amount of groundwater entering a river, affecting the amount and quality of instream habitat (Falke et al., 2011; Power, Brown, & Imhof, 1999) and the rates of biogeochemical processes in the hyporheic zone such as organic matter breakdown (Burrows et al., 2017). Vegetation composition and habitat quality of springs (Fensham & Fairfax, 2003; Powell et al., 2015) are also governed by temporal changes in groundwater storage, which affects groundwater discharge to these GDEs.

3.3.2 | Measuring GDE–groundwater connectivity

Many techniques exist for measuring GDE–groundwater connectivity, particularly in vegetation and riverine GDEs (Bertrand, Siergieiev, Ala-Aho, & Rossi, 2014; Eamus et al., 2015; Richardson et al., 2011). These methods include tracers, isotopes and geochemical techniques (Atkinson et al., 2015), physical techniques using data on hydraulic heads or from fine-scale radar (Huggenberger, Hoehn, Beschta, & Woessner, 1998; Lamontagne et al., 2014), methods comparing proportions of annual water use derived from groundwater (e.g., via evapotranspiration, Eamus et al., 2015), and statistical modelling or correlative techniques that infer GDE–groundwater connectivity from changes in ecological responses along groundwater gradients such as depth to water table (Elmore et al., 2006).

Additional to the above methods, which can be costly and resource intensive (e.g., for particular isotopes), GDE–groundwater connectivity in stream and hyporheic systems can be calculated using measures of streambed leakage coefficients and the differences between stream stage and aquifer heads (Hiscock & Bense, 2014). When the head difference between stream stage and aquifer is positive, there is likely to be a connection and, given suitable conditions of hydraulic conductivity, groundwater flows from the aquifer to the stream and hyporheic zone. Groundwater flows into streams and hyporheic zones may be nonlinear, with possibly rapid increases in groundwater flow for small head changes when head difference is small, as well as limits on the amount of flow as the head difference becomes larger (Rushton & Tomlinson, 1979).

We acknowledge that the above techniques may require information that is not readily available, particularly at landscape or management relevant scales (e.g., entire catchments). In such cases, statistical modelling approaches (e.g., Elmore et al., 2006) could be used to estimate GDE–groundwater connectivity at broad scales. Coupling satellite measurements of ecological responses (e.g., MODIS; Kath et al., 2015) and groundwater (e.g., GRACE; Richey et al., 2015) could also be used to estimate GDE–groundwater connectivity over large areas. Finally, interpolation and hydrogeological modelling techniques could be used to interpolate GDE–groundwater connectivity across landscapes. Interpolation techniques have been successfully applied in climate sciences (Jeffrey, Carter, Moodie, & Beswick, 2001) and could be similarly applied to estimate GDE–groundwater connectivity over broad scales when information is scant.

Ecohydrological links between GDE–groundwater connectivity, groundwater regime, and different GDEs can be explored by using a hydrogeograph to compare the distribution of GDEs with their measured levels of GDE–groundwater connectivity in a given area (Figure 5). This allows relationships between groundwater regime and multiple GDEs to be visualized together, illustrating how different (and potentially linked) GDEs become connected or disconnected to groundwater under different groundwater-regime scenarios. Using this approach, statistics on the components of GDE–groundwater connectivity can be determined and related to indicators of ecological responses for suites of GDEs through time and/or space.

Uncertainty around the measured levels of GDE–groundwater connectivity should also be ascertained. This uncertainty includes inherent hydrological variability, error in groundwater-regime measurements, and uncertainty in relationships between ecological indicators and different components of the groundwater regime. For example, Kath et al. (2014) quantified relationships between forest condition and groundwater depths and suggested that at the 90% confidence level across the studied catchment, riparian forests were connected at groundwater depths of 12.5–20.8 m. Quantifying uncertainty in GDE–groundwater connectivity enables risk-based assessments of the potential consequences of groundwater regime alteration, a key application of FERGRA (see later). Large uncertainty in measures of GDE–groundwater connectivity might indicate that more data need to be collected or different approaches for assessing connectivity are needed.

3.3.3 | Case study Step 3: Assessing GDE–groundwater connectivity in the Condamine catchment

Data on GDE–groundwater connectivity are unavailable for the Condamine catchment in the vicinity of our example bore. Therefore, we generated hypothetical values for GDE connectivity for each of our conceptual GDE examples, based on groundwater depth, of −6.5 m for the stream, −7 m for the hyporheic zone, and −30 m for the aquifer. GDE connectivity levels can range from less than 10 cm (Aldous & Bach, 2014) to many tens of meters for aquifers supporting stygofauna; Halse et al. (2014) recorded animals to a depth of 88 m. Our estimate of GDE–groundwater connectivity for riparian vegetation is from Kath et al. (2014), who estimated riparian vegetation (*E. camaldulensis*) was connected to groundwater at depths between 12.5 and 20.8 m. In this case study example, we use the value of 18 m to fit within this estimated range.

Having estimated GDE–groundwater connectivity for each GDE class, we derived statistics that describe each GDE's relationship with the groundwater regime in each 5-year time segment shown in Figure 6. Aquifer GDEs were permanently connected to groundwater, with frequency of connection stable across all periods assessed. In contrast, stream and hyporheic zone GDEs showed greater variation in frequency and duration of connection (shown as intersections of the hydrogeograph with horizontal lines on Figure 6). For example, at a GDE–groundwater connection level of −6.5 m, the stream GDE was only connected to groundwater in the periods 1975–1979 (92% of months), 1980–1984 (45% of months), and 1985–1989 (5% of months). The hyporheic zone GDE, with a GDE–groundwater connection level only 0.5 m lower than the stream GDE, was connected in more years (1970–1974 and 2010–2014, Figure 6) and for a longer total period of time (30% vs. 16%). In this conceptual example of the Condamine catchment, the hyporheic zone GDE is connected to groundwater almost twice as long as the stream GDE, illustrating the potentially important effect on GDE–groundwater connectivity for different GDE classes caused by small variances in groundwater depth. This conceptual example is corroborated by field data from different classes of GDEs (e.g., Aldous & Bach, 2014; Elmore et al., 2006; Falke et al., 2011).

Across the seven riparian sites (see Case-study Steps 1 and 2 for data details), groundwater depths ranged from −24.0 to −17.7 m, and frequency of connection to groundwater ranged from 0 to 100%, with a mean of 35%. The most negative rate of groundwater depth change was −0.6 m yr^{-1}; the most positive was 0.5 m yr^{-1}. Overall, there was a persistent decline in groundwater levels, with a collective rate of change across all years and sites of −0.1 m yr^{-1}. R code for calculating each of these statistics and other components of GDE connectivity is in Appendix S1.

3.4 | Step 4: Developing and testing hypotheses to assess GDE ecological responses to groundwater regime alteration

Step 4 of FERGRA integrates the information on the groundwater regime, extent of alteration, and the effects on various components (e.g., timing, duration, and frequency) of GDE–groundwater connectivity from Steps 1–3 to derive and then test hypotheses about ecological responses of different GDEs in a landscape to groundwater regime alteration. Ideally, these hypotheses should include the likely mechanism underlying the proposed relationship between the predicted GDE responses and groundwater regime alteration (cf. Poff et al., 2010) and specify alterations in one or more components of GDE–groundwater connectivity resulting from groundwater regime alteration. Table 2 presents some examples of these hypotheses and their proposed underlying mechanisms. Although these examples only refer to single components of GDE–groundwater connectivity, more complex hypotheses about interacting components could be derived. Furthermore, groundwater quality is also relevant because the predicted ecological responses to altered groundwater regime and GDE–groundwater connectivity may be very different between, for example, fresh and saline groundwater (e.g., Menció et al., 2014).

Various studies have developed hypotheses to test the existence of threshold responses of groundwater-dependent vegetation to changes in depth to groundwater. For example, Elmore et al. (2006) hypothesized alkali meadow vegetation would cease to respond to changes in water table beyond a certain depth, concluding that vegetation cover only responded to precipitation when groundwater depths declined below a threshold of 2.5 m. Another study by Aldous and Bach (2014) tested hypotheses about the effects of groundwater drawdown on wetland plants and peat accretion in fens in central Oregon. They inferred that maximum depths to the water table of 0.9 to 34.8 cm for fen plants and 16.6 to 32.2 cm for peat accretion can be tolerated in these GDEs. However, we are not aware of any published studies that have proposed and tested hypotheses about the ecological responses and their underlying mechanisms to altered groundwater regime and GDE–groundwater connectivity for multiple different GDEs in the same landscape concurrently (cf. different species within the same GDE: Scott, Shafroth, & Auble, 1999; Sommer & Froend, 2014).

Testing hypotheses about mechanisms of GDE–groundwater connectivity in different GDEs concurrently would facilitate extrapolation to similar landscapes with limited data or under future scenarios of altered groundwater regime. In the Akiraree River basin, Colorado, Falke et al. (2011) modelled changes in groundwater connections (using water table levels) and streamflow to predict refuge pool availability for fishes. Under the most conservative scenario of groundwater extraction, only 57% of dry-season refuge pools would persist by 2045. Although these authors did not assess responses of fishes to groundwater extraction, their inference that extraction altering groundwater regimes would reduce groundwater connections to streams and therefore reduce dry-season refuge habitats follows FERGRA's logic in identifying potential mechanisms and posing testable hypotheses.

Again, there are some major challenges to be considered when testing hypotheses about different GDEs' ecological responses to groundwater regime alteration. First, GDEs in the landscape are often affected by other stressors such as surface-water regime alteration (e.g., King et al., 2015) and land uses such as grazing (Powell et al., 2015). Disentangling these effects from groundwater regime alteration is difficult but data on changes in surface water and land use can be incorporated into models of GDE responses (e.g., Korbel & Hose, 2015) to help establish potential sources of variance. Second, most tests of these hypotheses require some knowledge of antecedent conditions and likely lag effects (Alley et al., 2002). Antecedent conditions may be inferred from long-term hindcasting and/or isotope approaches that can indicate long-term changes in groundwater systems (Green et al., 2011). Lag effects can be incorporated into

TABLE 2 Conceptual examples of hypotheses about likely ecological responses in specific GDEs to changes in a component of GDE–groundwater connectivity and the likely underlying mechanisms

GDE–groundwater connectivity component	Example hypotheses and mechanisms
Duration	Hypothesis: A decline in the duration of connection between a stream GDE and groundwater will correspond with a decline in stream macroinvertebrate richness. Mechanism: Where groundwater contributes to the persistence of refuge pools and prolongs the duration of stream flow in a groundwater-fed dryland stream, more species of stream macroinvertebrates will persist than in nearby rivers that flow for shorter periods and/or dry completely.
Frequency	Hypothesis: An increase in the frequency of groundwater connections to the hyporheic zone will correspond with higher rates of hyporheic organic matter decomposition. Mechanism: Breakdown of hyporheic organic matter such as buried leaf litter is faster in saturated than unsaturated conditions because this microbially mediated ecosystem process is accelerated by sustained high moisture levels and the more stable environmental conditions provided by groundwater inputs; less frequent groundwater-hyporheic GDE connections increase variability in environmental conditions, reduce interstitial moisture levels and interrupt organic matter decomposition by microbes.
Timing	Hypothesis: Greater variability in timing of groundwater connections will correspond to a reduction in EVI data from groundwater-dependent riparian forests in the Condamine catchment. Mechanism: A highly variable water table induces broader root profiles and shallow mean root depths, because roots remain shallow to avoid oxygen depletion when water table level is high. In contrast, a low variability water table induces root growth into deeper layers thus facilitating greater groundwater access for vegetation that prevents decline in condition (e.g., as indicated by EVI), especially under drought.
Magnitude	Hypothesis: Stygofauna richness will be negatively correlated with large, rapid negative rates of change in groundwater depth. Mechanism: A large, rapid fall in groundwater depth potentially strands and kills some groups of stygofauna because the water table recedes too rapidly for invertebrates to track, and some taxa cannot survive 48 hr in dry sediments. Reductions in the magnitude of groundwater connection also reduce the extent of available habitat, further reducing stygofauna richness.

Note. Results from testing these hypotheses in the Condamine catchment case study are presented in Table 3. GDE: groundwater-dependent ecosystem; EVI: enhanced vegetation index.

ecological modelling of GDE responses to altered groundwater regimes. For example, Kath et al. (2015) modelled forest responses to groundwater depth and salinity change with a 6-month lag. Finally, there are the challenges posed by the ubiquitous mismatches in scale between hydrogeological and ecological data as well as the multiple sources of uncertainty (discussed earlier). Uncertainty and downscaling techniques being developed in climatology and hydrogeology (e.g., Crosbie et al., 2011; Green et al., 2011) show potential for providing more spatially relevant groundwater data for GDE analyses.

3.4.1 | Case study Step 4: Hypothesizing and testing ecological responses to groundwater regime alteration in the Condamine catchment

Having linked our hypothetical GDE responses to our groundwater regime data using measures of GDE–groundwater connectivity, we can now propose and test different hypotheses on the basis of the conceptual examples introduced in Steps 2 and 3. The first hypothesis focused on the component of duration of GDE–groundwater connectivity and was that a decline in the duration of connection between the stream GDE and groundwater would correspond with a decline in stream macroinvertebrate richness (Table 2). The underlying mechanism is that where groundwater contributes to the persistence of refuge pools and prolongs the duration of stream flow, more species of stream macroinvertebrates will persist than in nearby rivers that flow for shorter periods and/or dry completely (Table 2). Many studies have demonstrated a positive correlation between duration of flow and stream macroinvertebrate richness (review in Stubbington et al., 2017). However, when we tested our hypothesis using simple linear regression of our simulated data, the association was nonsignificant (Table 3) and we rejected our hypothesis. As in many dryland rivers, taxonomic richness of macroinvertebrates along the Condamine River is low but variable (Chessman et al., 2010; Marshall et al., 2006), potentially limiting the resolution of our analysis to detect significant differences associated with flow duration. Alternatively, groundwater inputs may be less relevant in the Condamine than in other rivers (e.g., Stubbington et al., 2011) so that reliance of this stream GDE on groundwater is low or transitory.

The second hypothesis addressed the frequency of GDE–groundwater connectivity and its effect on a fundamental GDE process, organic matter decomposition. We hypothesized that an increase in the frequency of groundwater connections to the hyporheic zone would correspond with higher rates of hyporheic organic matter decomposition (Table 2). This has been demonstrated in several eastern Australian hyporheic zones where groundwater inputs increased leaf litter breakdown by 48% and cotton-strip decomposition by 124% compared with rates in the overlying surface environments (Burrows et al., 2017). The proposed mechanism for the faster breakdown of hyporheic organic matter in saturated conditions provided by groundwater is that this microbially mediated ecosystem process is accelerated by sustained high moisture levels and the more stable environmental conditions, both of which would be disrupted by increased frequency of disconnections between the GDE and the groundwater (Table 2). When we tested this hypothesis with simulated data from our conceptual example, the association was nonsignificant

TABLE 3 Results for the four conceptual examples presented in Table 2

Hypotheses	Parameter tested	Parameter value (standard error)	Significance value	Accept/reject hypothesis
1. A decline in the duration of connection between a stream GDE and groundwater will correspond with a decline in macroinvertebrate richness	Duration of groundwater connection	−0.063(0.069)	0.388	Reject at *P* = 0.05
2. An increase in the frequency of groundwater connections to the hyporheic zone will correspond with higher rates of organic matter decomposition	Frequency of groundwater connection	0.359(3.004)	0.908	Reject at *P* = 0.05
3. Greater groundwater variability (calculated as variance) will correspond to negative change in EVI.	Variability (variance) in groundwater depth	−0.067(0.016)	0.0007	Accept at *P* = 0.05
4. Stygofauna richness will be negatively correlated with large, rapid negative rates of change in groundwater depth.	Magnitude and rate of change in groundwater depth	147.867(32.746)	0.003	Accept at *P* = 0.05

Note. GDE: groundwater-dependent ecosystem; EVI: enhanced vegetation index.

(Table 3). This lack of support for our hypothesis may be for several reasons, including the overriding effects of other hyporheic variables such as fine sediment size and low dissolved oxygen concentrations on organic matter breakdown (Boulton, Datry, Kasahara, Mutz, & Stanford, 2010) in the Condamine River, mismatches between the hydrogeological and ecological scales of measurements in the conceptual example, and high inherent variability in organic matter breakdown when saturation levels fluctuate because of variable frequencies of groundwater connectivity.

Our third hypothesis, exploring the influence of timing of GDE–groundwater connectivity, was that greater variability in timing of groundwater connections would correspond to a reduction in EVI from riparian forest GDEs in the Condamine catchment (Table 2). The proposed mechanism is that as groundwater variability influences the physical characteristics of vegetation (e.g., rooting structure and depth), we would expect riparian forest access to groundwater to differ with groundwater variability (Merritt, Scott, Poff, Auble, & Lytle, 2010; Tron, Laio, & Ridolfi, 2014). Specifically, as high groundwater variability promotes shallow rooting depth to avoid the lack of oxygen that occurs when the water table is high, we expect riparian forests to have better access to groundwater and to be in better condition (as indicated by EVI) at low groundwater variability sites that allow deeper rooting depths (Tron et al., 2014). Unlike the other conceptual examples, this hypothesis was tested with actual data from the Condamine catchment, using a generalized linear mixed model to deal with spatially and temporally clustered data (Bolker et al., 2009). There was a significant association (Table 3), indicating that increased variability in connection between the groundwater and this GDE is associated with a negative change in riparian vegetation condition. Water table variability has been demonstrated to be a key determinant of the distribution and condition of a range of floodplain and riparian vegetation (Johansson & Nilsson, 2002; Leyer, 2005; Merritt et al., 2010). Vegetation biomass, root depth, and root architecture are structural traits of plants responsive to hydrologic variability (Merritt et al., 2010), and so it is expected that these attributes would indicate changes in groundwater variability, as was the case for our remotely sensed data on riparian forests in this case study example.

The final hypothesis assessed the component of magnitude and rate of change on stygofauna richness in a confined aquifer and arose from experimental evidence (Stumpp & Hose, 2013) that some groups of stygofauna are adversely affected by rapid rates of groundwater drawdown and may not survive 48 hr in drying sediments. We hypothesized that stygofauna richness would be negatively correlated with large, rapid negative rates of change in groundwater depth (Table 2). The mechanisms underlying the predicted ecological response by stygofauna to large and rapid declines in water table are the direct effects of stranding and subsequent desiccation on some taxa and the indirect effects of reduced available habitat and resources for other taxa (Table 2). Testing this hypothesis with simulated data from our conceptual example yielded a significant association (Table 3), indicating that a large, rapid fall in water table corresponds with declining stygofauna richness in this hypothetical aquifer in the Condamine catchment.

3.5 | Step 5: Using FERGRA to guide GDE management and decision making

Collating data on alterations of groundwater regime and GDE–groundwater connectivity followed by tests of hypotheses about their potential ecological effects (Steps 1–4 in FERGRA) informs researchers and managers about how multiple different GDEs respond to groundwater regime alteration in a landscape. The final step in FERGRA is to use this information to estimate how much groundwater regime alteration can take place to meet socio-economic demands such as groundwater extraction before environmental values and ecosystem services associated with GDEs are compromised beyond some acceptable level. Acceptable limits have been suggested for some specific GDEs (e.g., Aldous & Bach, 2014; Perkin et al., 2017), but far more work is needed. In particular, it is essential to identify thresholds of the various components of groundwater connectivity (e.g., duration and frequency) for GDEs that, if exceeded by groundwater regime alteration, would lead to precipitous declines in the provision of one or more GDE ecosystem services in a given landscape. FERGRA provides a logical framework to systematically determine relationships, particularly thresholds, of different GDE's responses to altered groundwater regimes so that managers can integrate these relationships with acceptable limits to derive appropriate GDE management strategies at a landscape or catchment level.

Using this information on ecohydrological relationships and acceptable ranges of values for key attributes (e.g., specific taxa, biodiversity, and ecosystem processes) of different GDEs, managers can then identify whether groundwater regimes and GDE–groundwater connections need to be either maintained or restored (Figure 7a). Maintaining groundwater regimes within a range that can support key ecological attributes of GDEs is preferable because restoration is more costly and less effective than preventing ecological degradation (Hobbs, Hallett, Ehrlich, & Mooney, 2011). Indeed, restoration might not be feasible in some GDEs or be severely hampered by the slow response times of aquifers and the high dependence of many socio-economic activities on groundwater use (Gleeson et al., 2010). Even for aquifers where relatively rapid recovery of natural groundwater regimes may be possible, there is scant research about whether it is possible to restore multiple GDEs in a landscape by reversing groundwater regime alteration.

Having decided whether to maintain or restore a suite of GDEs, appropriate management strategies must be identified. In general, this means changing the spatial (e.g., buffering distances between extraction and GDEs) and/or temporal (e.g., timing and magnitude of extraction) dynamics of current groundwater use activities (Eamus & Froend, 2006). There are numerous tools and optimization techniques for making trade-offs between socio-economic and environmental demands linked to groundwater regime alterations (e.g., Reed & Minsker, 2004).

3.5.1 | Case study Step 5: Using FERGRA to guide GDE management in the Condamine catchment

Testing the four hypotheses in Step 4 of FERGRA identified rate of change and variability in groundwater depth as potentially important components of the groundwater regime for stygofauna taxonomic richness and riparian forest condition, respectively, in the Condamine catchment example. The fifth step in FERGRA uses this information and appropriate ecohydrological modelling to help decide between two broad options (Figure 7a) to manage relevant components of the groundwater regime for these two types of GDEs. If researchers and managers agree that an acceptable level of stygofauna richness in this region is six or more taxa, rates of groundwater change should be managed to avoid declines faster than ~0.01 m month^{-1} (Figure 7b). Around this acceptable level, there is also likely to be some error or uncertainty, represented by the pale grey rectangles in Figure 7. When values are in this area, it could represent a caution zone or serve as trigger for pre-emptive management actions to prevent further undesirable change. Similar reasoning applies for the riparian forest GDE; if zero change in riparian condition (as indicated by EVI) is set as the acceptable level (again, with an associated error level), then variability in groundwater depth should be maintained at or below ~1 m^2 or less (Figure 7c).

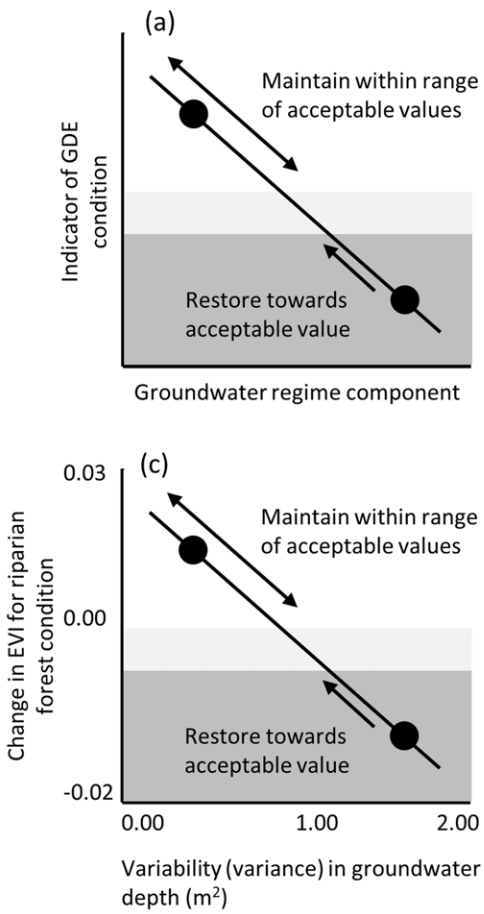

FIGURE 7 (a) When an ecohydrological relationship has been established between a given indicator of GDE condition and one or more components of the groundwater regime, managers need a threshold and error range of acceptable values of the indicator (pale grey box) to help them decide between management strategies. Values in this range (pale grey box) could also represent a "caution zone" and/or serve as trigger for pre-emptive management actions to prevent further undesirable change. Conceptual examples are shown of the application of this approach to (b) stygofauna richness associated with rate of change and (c) riparian forest condition (as rates of change in enhanced vegetation index [EVI]) associated with groundwater variability. See text for more details

In this conceptual example of the Condamine catchment, stygofauna within the aquifers can hypothetically be managed by ensuring groundwater extraction rates do not lower depths to groundwater faster than −0.01 m per month, whereas the condition of riparian forests can be managed by smoothing groundwater extraction rates to reduce the variability in groundwater depth below the threshold value of 1 m^2. There are various ways that groundwater regime could be managed to achieve this. For example, Eamus and Froend (2006) suggest that modifying the timing, magnitude, and rates of groundwater drawdown to avoid times of peak environmental demand substantially reduces risks to GDEs and may allow some groundwater-dependent biota to adapt to lower water tables. Using FERGRA to concurrently assess ecological responses by a suite of GDEs to groundwater regime alteration enables managers to seek strategies that optimize conditions for multiple different GDEs in a landscape rather than only one or two types of GDEs.

4 | GOALS AND POTENTIAL EXTENSIONS OF FERGRA

The main goal of FERGRA is to provide a logical, consistent, and robust framework for guiding collective management of multiple different GDEs in a landscape experiencing alteration in its groundwater regime. Sustainable management of the full suite of GDEs in a landscape is essential for maintaining biodiversity and ecosystem services linked to human well-being (Griebler & Avramov, 2015). Most current approaches to GDE management tend to focus on a single class of GDEs. Particular focus has been on groundwater-dependent vegetation (e.g., Aldous & Bach, 2014), probably because ecohydrological relationships between components of the groundwater regime (usually depth to groundwater) and vegetation condition of this class of GDEs is best known. However, threats from groundwater alteration may be more severe for other types of GDEs in the same landscape (e.g., discharge springs; Powell et al., 2015), and so a framework such as FERGRA that addresses multiple different GDEs concurrently is needed.

A second goal of FERGRA is to organize the currently fragmented research on GDEs so that knowledge gaps can be identified and prioritized. FERGRA's applicability across diverse GDE types counters the current problem where most research and management has been limited to single GDEs, leading to isolated “silos” of knowledge and assumptions (Parsons, Caruso, Barber, & Hayes, 2011). Comparison of ecohydrological relationships of multiple different GDEs across a landscape is rare; researchers typically focus on one GDE type at a time (e.g., vegetation [Aldous & Bach, 2014; Eamus et al., 2015], aquifer biota [Hancock & Boulton, 2009; Korbel & Hose, 2015], and stream biota [Brunke et al., 2003; Falke et al., 2011]).

Examining groundwater regimes and GDE–groundwater connectivity for multiple (and often interlinked) GDEs simultaneously could better foster a landscape perspective, largely lacking in current GDE research, where groundwater systems are viewed as an interacting mosaic in the “GDE-scape.” For example, embracing the heterogeneity that exists within and across streams by viewing them as a riverscape has been successful in stream ecology, revealing important knowledge gaps in the interactions of subsystems such as the riparian zone, stream channel, and underlying sediments (Wiens, 2002). The currently limited information about GDEs and hydrogeology at commensurate scales is likely to constrain research in the near term. However, as information about groundwater regimes and ecohydrological relationships of GDEs improves over the longer term, knowledge gaps about interactions and linkages between GDEs at landscape scales could be addressed using FERGRA because this collective approach is not specific to a single GDE type.

FERGRA has some potential extensions too. One is the integration of the framework into various GDE classifications that are presently in use. For example, work within the European Water Framework Directive has developed multiscale classifications for GDEs (Bertrand, Goldscheider, Gobat, & Hunkeler, 2012) that, integrated with FERGRA, have potential to aid management of GDEs in data-poor areas. Further, the extension of FERGRA to widely used GDE classifications could generate crucial broadscale information about GDEs that may be the most sensitive to groundwater regime change, as well as those aspects of the groundwater regime (e.g., rate of change and magnitude) that are likely to be the most influential on GDE biota and ecosystem processes.

FERGRA could also be applied within adaptive management groundwater monitoring and policy frameworks to guide decision making when there is scant information on GDEs (Rohde, Froend, & Howard, 2017). This would especially apply in situations where inadequate data prevent quantification of changes (see Premise 5) and force immediate management decisions to be based on qualitative assessments (e.g., categories of risk). However, although initial decision making might depend on qualitative assessments, managers could use FERGRA within an adaptive management framework to identify the key components of the groundwater regime and GDE responses in need of quantification and adjust their decision making as better information becomes available. For example, in Australia, some states utilize adaptive and risk management approaches to mitigate undesired impacts on potential vulnerable GDEs in the interim until better information becomes available (Parsons et al., 2011; Rohde et al., 2017).

Another potential extension of FERGRA is the application to integrated water resource management of both surface and groundwater regimes to maintain environmental values and ecosystem services. Currently, many ecologically based water resource management frameworks focus on surface-water flow regimes (e.g., Poff et al., 2010), and environmental flow management in rivers struggles to account for the importance of surface–groundwater interactions and of aquifer and vegetation GDEs (Parsons et al., 2011). Extending FERGRA to complement frameworks focused on surface water facilitates inclusion of multiple GDEs and their water requirements into fully integrated water resource research and management that explicitly addresses the dynamics and interactions of both surface water and groundwater.

5 | CONCLUSIONS

Groundwater extraction has numerous socio-economic benefits but alters groundwater regimes, impairing other ecosystem services provided by GDEs (Griebler & Avramov, 2015). Increasing societal awareness of the importance of GDEs and their services (Boulton, 2009) demands more collective research and management (Griebler & Avramov, 2015; Palmer, Hondula, & Koch, 2014), especially of multiple linked GDEs in the landscape. Estimates of the hydrological and ecological changes to the full array of GDEs caused by groundwater regime alteration are needed to inform decisions about acceptable levels of groundwater extraction to preserve the full range of values and ecosystem services provided by GDEs.

The framework proposed here addresses this core issue of quantifying and understanding hydrological and ecological changes to GDE-linked ecosystem services caused by groundwater regime alteration and for guiding collective management of multiple different GDEs in the landscape concurrently. Global-scale analyses of threats to the ecosystem services of surface freshwaters (Green et al., 2015) bemoan the paucity of data and understanding about aquifers and other GDEs, especially the impacts of altered groundwater regimes. FERGRA shows great promise for collating what we know and do not know about GDEs, using a hypothesis-based approach to seek generalizations about GDEs' ecological responses to groundwater regime alteration and adopting a collective perspective across the diverse array of GDEs in the landscape concurrently to fully integrate management of surface and ground waters.

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