Defining renewable groundwater use to improve groundwater management

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Abstract

Groundwater systems are commonly but variously defined as renewable or non-renewable based on natural fluxes of recharge or on estimates of aquifer storage and groundwater residence time. However, the principle of capture challenges simple definitions: groundwater pumping can alter the rates of groundwater recharge and/or discharge over a characteristic groundwater response time related to the hydraulic properties of the subsurface, the length scales of the system and the relative position of the abstraction wells to the hydraulic boundaries. If pumping exceeds the possible capture, continued depletion or the recovery time will then be more strongly determined by the relative rates of pumping and recharge. Hence, we argue here that a groundwater system cannot be renewable or non-renewable in and of itself, but only with reference to how the groundwater is being used. We propose a new definition explicitly focusing on use: *renewable groundwater use allows for dynamically stable re-equilibrium of groundwater levels and quality on human timescales (~50-100 years)*. The definition combines both 'flux' and 'storage' based perspectives on renewable use. Further, we show how a matrix of combinations of (1) the ratio of pumping to possible capture along with (2) the response or recovery timescales implicit in this definition, leads to a useful four-quadrant framework for groundwater management. The quadrants, illustrated using case studies from aquifers around the world, are a practical tool for quantitatively assessing the physical limits to the sustainability of a pumped groundwater system alongside requisite environmental and social aspects and potential pathways towards greater sustainability.

1. Motivation

As the largest store of accessible freshwater on Earth, groundwater is under increasing pressure as a resource, currently underpinning a large proportion of irrigated agriculture worldwide and being the source of drinking water for around 2 billion people (Famiglietti, 2014). Over-abstraction of groundwater can harm groundwater-dependent ecosystems and cause groundwater salinization, increased frequency and severity of hydrological drought, land subsidence, and sea-level rise (Bierkens and Wada, 2019). Clear conceptual thinking and robust quantification of groundwater management options is therefore a pressing global problem within the context of groundwater sustainability. However, it is increasingly clear that existing definitions and metrics of large-scale groundwater use are inconsistent and confusing which can undermine this goal (Ferguson *et al.*, 2020; Gleeson *et al.*, 2020).

Here we aim to define more clearly the notion of renewable groundwater, and how this concept fits within a broader sustainability framework for groundwater management. This is not just a semantic exercise: identifying paths toward groundwater sustainability depends on defining clear goals, and a clarification of terms is important to resolve existing inconsistency and confusion between other scientists, managers, policy makers and stakeholders.

To begin, we address the question of whether groundwater is in fact a renewable resource, and use a simple thought experiment to propose a new definition of renewable groundwater use. We then show how this leads naturally to a four quadrant framework for quantitative groundwater resource management and illustrate this with reference to case studies from aquifers around the world. Finally, we discuss how renewable groundwater use fits within a broader understanding of groundwater sustainability and how the quadrant framework presents challenges and opportunities for further developments to map sustainability pathways.

2. Is groundwater a renewable resource?

We first question whether the common assumption that groundwater is a renewable resource is a sound one. It is instructive to begin by considering how a 'renewable resource' can be defined in more general terms as:

> *"A natural resource (such as fresh water, a forest, or renewable energy) that is replaced at a rate which is at least as fast as it is used, which has the ability to renew itself and be harvested indefinitely under the right conditions, but which can be converted into a non-renewable resource if subject to overexploitation."*
>
> (Park & Allaby, 2017)

Hidden in a general definition of renewable natural resources such as this, is an implicit amalgamation of two distinct modes of resource use. First, there is the direct capture or abstraction of a *flux* of energy or matter. For example, this is the case for solar or wind energy which, while they may be used in conjunction with energy storage devices, essentially exploit the flux of photons or wind and convert them to other forms of useable energy at the point of capture. Renewability of some surface water resources can also be conceived in this 'flux-based' way, for instance where the capture of streamflow flux is used directly (e.g. for irrigation) or stored artificially for later use (e.g. in a reservoir). In contrast, other natural resources have a much greater propensity for natural *storage*, and it is predominantly the 'stock' of the resource which is allowed to accumulate and then be harvested or abstracted for human use. For example, such 'storage-based' definitions of renewability may be applicable in this way to forestry or fisheries, or large-volume surface water bodies such as lakes.

Groundwater flows, albeit slowly, but the recharge fluxes involved cannot be captured directly as for other 'flux-based' natural resources. Rather the groundwater must be pumped out of the ground at discrete points in a landscape via wells which are often very expensive to drill and maintain. Hence, while it may be tempting to define groundwater renewability using a 'storage-based' approach (Taylor, 2009), the infrastructure required for the abstraction of groundwater is immobile (for example, in comparison to forestry or fishery harvesting). Furthermore, as we elaborate further below, the timescales of groundwater storage depletion are not just determined by the rate of use, and can span several orders of magnitude. Hence, groundwater does not easily fit into either (flux-based or storage-based) categorisation and renewable groundwater has therefore previously been defined in several contradictory ways (Figure 1).

As outlined in detail by Gleeson *et al.*, 2020, flux-based definitions equate renewable groundwater with the rate of (pre-pumping) natural recharge (Döll and Fiedler, 2008; Wada *et al.*, 2010). This is a potentially useful and clear definition but will be inevitably incorrect, equating to the minimum likely capture, unless it includes the principle of increased recharge occurring during capture (Theis, 1940; Lohman, 1972). Capture, as we show in the next section, is a fundamental process to include in considerations of renewability. Flux-based definitions of renewable groundwater are convenient for integrated water resources management since they are readily combined with renewable surface water resources which are almost always defined using streamflows (Vörösmarty *et al.*, 2000; Alcamo *et al.*, 2003). However, when combined, they must not be double counted, and the overlaps between internally generated recharge and streamflow must be carefully accounted for (Food and Agriculture Organization of the United Nations, 2003).

Storage-based approaches, on the other hand, define renewable or non-renewable groundwater using a threshold value or mean renewal time, normally defined as the ratio of groundwater storage to the (pre-pumping) recharge rate (Margat et al. 2006). Again, in this definition, the principle of capture is often ignored but, more problematically, the logic of defining renewability in this way may be questionable from a hydraulic perspective. For example, it implies that groundwater in a larger aquifer receiving the same recharge flux as a smaller aquifer is inherently less renewable. This appeals to the commonly held perception that older groundwater may be less sustainable, but as has been outlined elsewhere, this is not a coherent position: both old and young groundwaters can be used in physically sustainable or unsustainable ways (Ferguson *et al.*, 2020).

Irrespective of the relative merit or coherence of the flux and storage-based definitions, the two definitions are inconsistent with each other since each implicitly excludes the other. This has led to recent attempts to harmonise the approaches. For example, Bierkens & Wada (2019) proposed that non-renewable groundwater is: "groundwater withdrawn that is not expected to be recharged on human time scales (>100 years)". Gleeson *et al.*, 2020 attempted to harmonise the flux-based and storage-based definitions defining renewable groundwater as: "any groundwater that can be dynamically captured during pumping that leads to a new dynamically stable equilibrium in groundwater levels within human timescales (∼100 years)". In this definition renewable groundwater is still seen as a property of the system itself, implicitly putting 'blame' for non-renewability on the groundwater, not on the user (Figure 1).

In summary, determining a useful definition of renewable groundwater is more complex than it is for many other natural resources. Our view is that existing definitions currently make it a potentially cumbersome at best, or misleading at worst, concept in the context of sustainability science. We propose here that renewable groundwater use may be more clearly and practically defined and that doing so can also lead to new insights into strategies for quantitative groundwater management.

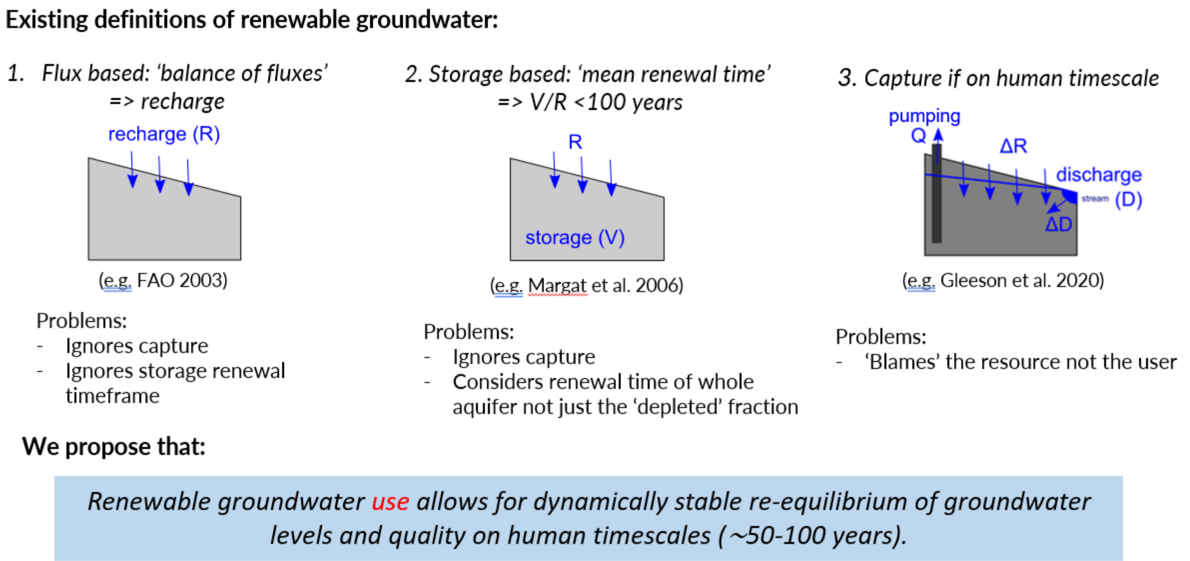

Figure 1. *Evolving definitions of groundwater renewability. We develop our proposed definition in Section 4 below.*

3. What happens when we pump?

Groundwater Quantity Considerations

To move towards a robust definition of renewable groundwater use, we now use a thought experiment of a simple aquifer and consider changes to its overall water balance with and without groundwater pumping. The groundwater balance of the aquifer can be stated as:

$$\frac{dS}{dt} = R - D - Q \qquad (1)$$

where S is total groundwater storage [L], t is time [T], R is the recharge rate [LT^{-1}], D is the rate of groundwater discharge other than pumping [LT^{-1}] and Q is the rate of pumping [LT^{-1}]. For the following discussion, it is useful to break–down these terms further as follows:

$$R = R_0 + \Delta R_Q + \Delta R_N \qquad (2)$$

$$D = D_0 + \Delta D_Q + \Delta D_N \qquad (3)$$

Where R_0 is the pre-development recharge rate, the change in the recharge rate from the pre-development rate due to pumping or natural transients is given by ΔR_Q and ΔR_N respectively, D_0 is the pre-development natural groundwater discharge rate, and the change in the discharge rate from the pre-development rate due to pumping or natural transients is given by ΔD_Q and ΔD_N respectively.

We assume the abstractions are widely spatially distributed, and hence all our considerations here are on the large-scale water balance, not the detailed hydraulics of an individual abstraction well, in a similar conceptual framework as (Bierkens, Sutanudjaja and Wanders, 2021). Further we assume that the aquifer has impermeable boundaries laterally and at its base, that it may receive recharge via precipitation and/or via stream leakage, and groundwater may also discharge naturally e.g. to streams as baseflow or as evapotranspiration. The recharge received by the aquifer reflects superposed climatic variations operating on multiple time periods, but is assumed to be in quasi-steady state and stationary over the time period of the groundwater development. i.e. in this thought experiment, $R_0 \sim D_0$ and $\Delta R_N \sim 0 \sim \Delta D_N$. Later we will also consider the implications of relaxing these assumptions.

In an unpumped situation, the hydraulic heads (hereafter "heads") in the aquifer fluctuate in a dynamic steady state around some value above the elevation of the lowest hydraulically-connected streambed overlying the aquifer (the 'drainage base-level'). If steady state pumping begins, groundwater is taken out of storage as the well drawdown cones superpose themselves across the aquifer. The evolving change in hydraulic gradients due to pumping causes a combination of (1) changing rates of recharge (ΔR_Q) through the capture of evapotranspiration (ET) or surface/shallow-subsurface runoff, (2) changing rates of discharge (ΔD_Q) through the capture of stream baseflow, evapotranspiration and/or altered rates of exchange with other neighbouring geological units. The combination of (1) and (2) are termed capture (C). i.e.:

$$C = \Delta R_Q - \Delta D_Q \qquad (4)$$

If the rate of pumping is less than the maximum possible rate of capture (C_{max}), the heads will gradually come to a new dynamic equilibrium (e.g. green and orange lines on Figure 2) at which time all of the pumping is being sourced from capture, and no longer from changes in storage. This maximum rate of capture is sometimes also referred to as the physically sustainable pumping rate (Bierkens and Wada, 2019; Gleeson *et al.*, 2020) - below this rate groundwater-surface water interactions remain bi-directional and at least part of the stream network will be perennial. If pumping then ceases, the reverse of the capture process occurs, with 'release' of baseflow, runoff and evapotranspiration ensuing until the system returns to a new dynamic steady state once more.

There may be some differences in the timescales of capture and release for example due to hysteretic relationships between heads via bi-directional stream-aquifer interactions. However, for the widely distributed pumping assumed here, both processes can be approximated using a characteristic timescale referred to variably as the reservoir co-efficient (Kraijenhoff van de Leur, 1958), aquifer response rate (Erskine and Papaioannou, 1997), time constant (Rousseau-Gueutin *et al.*, 2013), mean action time (Simpson, Jazaei and Clement, 2013), hydraulic response time (Alley *et al.*, 2002) or groundwater response time (GRT) (Cuthbert, Gleeson, *et al.*, 2019) which is the term we will use here. Such response times are principally related to the physical properties of the system in question (e.g. stream geometries, hydraulic properties of the aquifer, distance between geological and hydraulic boundaries) and not to the initial, pre-pumping, rate of recharge. Note, these hydraulic response times should not be equated or confused with groundwater residence times or ages (Ferguson *et al.*, 2020).

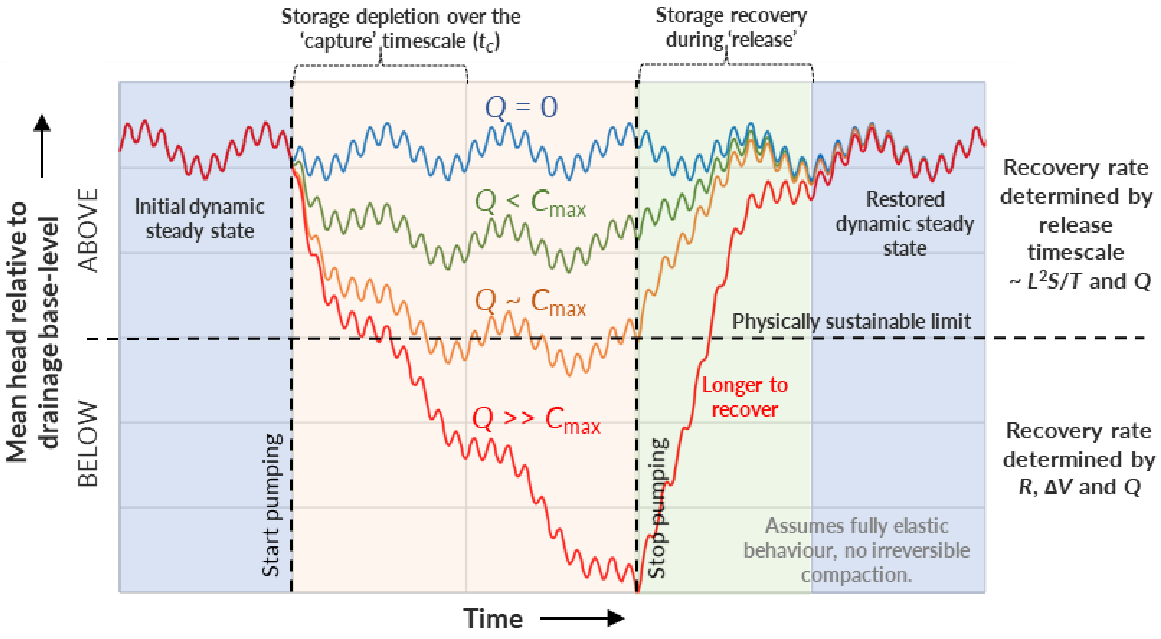

Figure 2. *Controls on the dynamics of pumping and recovery of a generic groundwater system. R = recharge, ΔV = storage depletion during pumping, Q = pumping, C_{max} = rate of maximum capture, L = characteristic aquifer length scale, S = storage coefficient, T = transmissivity. Plotted using a lumped explicit finite-difference solution to Equation 1, with $R = R_0$ defined by a superposition of sinusoidal signals of various periods, and D set as proportional to the head difference above the drainage base level while above the base level, and zero when below.*

Since the connection to surface water courses may vary with time for a given aquifer depending on the relative water level at a particular climatic state, the response time may also change through time. For an individual well, or group of wells, the time to full capture (Bredehoeft and Durbin, 2009) or release (t_C) may significantly deviate from the GRT of the aquifer as a whole depending on the relative position of the wells and boundary conditions. For example, shorter times to equilibrium and a greater proportion of depletion compared to capture are expected for wells positioned closer to streams (Konikow and Leake, 2014). Whether the full capture timescale is considered in relative or absolute terms is also an important consideration. Where the timescale is calculated as the relative drawdown (or streamflow depletion) as a proportion of the maximum drawdown (or streamflow depletion), the timescale is solely related to the hydraulic properties of the system and not the pumping rate. However, if the timescale is defined in absolute terms, for example with respect to a threshold change in groundwater level or streamflow, then the pumping rate will also be a contributing factor in the magnitude of the timescale for a new equilibrium to be established.

If the rate of pumping is greater than the maximum possible capture, the heads in the aquifer will eventually fall below the drainage base level (red line on Figure 2) and stream-aquifer interactions will become uni-directional. In this situation, recharge may still be received by the aquifer from losing streams but the aquifer is no longer able to hydraulically feedback on this process (Bierkens and Wada, 2019; Quichimbo, Singer and Cuthbert, 2020). In such cases the continued rate of lowering of heads, and the recovery of heads should pumping cease, will increasingly be governed by the relative rates of recharge and pumping in conjunction with the specific yield of the aquifer and the duration of the pumping. This rate of depletion or recovery thus has distinct controls from the lower pumping cases described above for situations where the groundwater is still able to feedback hydraulically to the stream network in a bi-directional sense. While heads are below the drainage base level, the rate of recovery will always be faster than the rate of depletion. This stems from the fact that during depletion the rate is determined by the difference between the recharge rate and pumping rate, whereas during recovery just the recharge rate becomes relevant. Longer recoveries are to be expected to result after longer pumping periods or higher pumping rates relative to rates of maximum capture. However, since there is a potentially complex trade-off between these two factors and the hydraulic properties of a particular aquifer, we do not propose quantitative metrics in a general sense here.

Similar thought experiments can be carried out for more complex aquifer systems. For example, there may be head-dependent inflows or outflows with other neighboring geological units, large water bodies, or the sea. In such cases there will be a more complex balance of capture with the landscape (via changes in streamflow and evapotranspiration) and laterally with the adjacent sources or sinks of groundwater. However, despite the more complex quantification of capture and release in practice, the overall concepts outlined above will still hold.

Groundwater Quality & Hydromechanical Considerations

We have focussed so far only on groundwater quantity, in line with the previous literature on renewable groundwater. However, capture resulting from pumping can also have significant consequences for other aspects which may impact groundwater renewability such as groundwater quality and hydromechanical changes in the aquifer.

Of particular importance is the situation when the quality of captured water is inconsistent with the background groundwater chemistry of the aquifer which the development is targeting. For example, contaminated or naturally more saline water may be induced to recharge an aquifer. This may happen where pumping moves the position of the saline interface either in coastal areas or island aquifers (Werner *et al.*, 2013), or where lower quality water is brought in from adjacent formations (Ferguson *et al.*, 2018). In some contexts pumping may also cause internal shifts in chemistry leading to deteriorations in groundwater quality, for example via the mechanism of "Anthropogenic Basin Closure and groundwater SALinization" (ABCSAL) (Pauloo *et al.*, 2021).

It is possible that if abstraction rates are high enough, the time taken for water quality to recover to its pre-pumping state may be much higher than the hydraulic response time. For example, with regard to saline groundwater, interfaces between saline and fresh water do not remain sharp since conductivity heterogeneity and isotropy and larger dispersivities may induce mixing, which results in brackish zones that may take very long times to disperse (see Bierkens and Wada 2019 and references on salinization therein).

There are also contexts in which the lowering of heads in an aquifer can cause irreversible changes in the properties or state of the system. Firstly, dewatering can cause consolidation of more compressible formations leading to land subsidence and irreversible (i.e. non-elastic) loss of groundwater storage (Galloway and Burbey, 2011). For example, the California Central Valley aquifer system permanently lost an estimated 0.4–3.25% of its storage capacity due to pumping during a period of drought between 2012 and 2015 (Ojha, Werth and Shirzaei, 2019). Secondly, although it is not yet well understood, it is possible that, as a result of non-linear dynamics in a coupled climate-vegetation-hydrological system, pumping may cause a groundwater system to switch states to a regime from which it will not recover simply by stopping pumping (Peterson *et al.*, 2021).

These considerations indicate that in certain contexts the physically sustainable pumping rate ($C_{\max}$) may be an overestimate of the maximum rate of pumping which also allows the aquifer to recover flows and storage of consistent quality groundwater, which we will term $Q_{\max}$ in the rest of the paper.

4. Redefining renewable groundwater use

The above thought experiment indicates that, while some degree of storage depletion is always required for any groundwater development (Theis, 1940), there are two major controls on the extent and timescales for aquifer depletion and recovery:

- the rate of pumping (Q) relative to the maximum capture ($Q_{\max}$) which amounts to a more restrictive definition of whether the pumped system is 'capture-constrained' or not (Konikow and Leake, 2014).
- the time to full capture or recovery (t_C) of pumping at the rate $Q_{\max}$, relative to a given human timescale of interest (t_H).

These factors relate directly to 'flux-based' and 'storage-based' definitions of groundwater renewability respectively, from previous definitions in the literature. By defining four quadrants based on combinations of these two controls we can more accurately describe groundwater renewability of each as follows:

a) **Renewable Use** (lower left quadrant): *systems where pumping is less than the maximum capture, and also have short response times ($Q < Q_{max}$, $t_C < t_H$).* From a flux-based perspective this quadrant therefore represents renewable groundwater use. From a storage-based perspective, the situation also always represents renewable use irrespective of how long the pumping continues. This is the case in this quadrant since $Q < Q_{\max}$ and thus the recovery time is solely controlled by the hydraulic response which will approximate to the t_C and thus will also be less than t_H.

b) **Non-flux-renewable Use** (lower right quadrant): *systems where pumping is greater than the maximum capture, but have short response times ($Q > Q_{max}$, $t_C < t_H$).* From a flux-based perspective, this is a situation of non-renewable groundwater use. However, from a storage-based perspective, it is possible for the situation to represent renewable use depending on the specific combination of pumping duration and magnitude, t_C and water quality considerations.

c) **Flux-renewable Use** (upper left quadrant): *systems where pumping is less than the maximum capture, but have long response times ($Q < Q_{max}$, $t_C > t_H$).* From a flux-based perspective, this is a situation of renewable groundwater use. In this quadrant, the recovery time will be controlled only by the hydraulic response time since $Q < Q_{\max}$ and groundwater levels will never fall below the drainage base level. Hence, from a storage-based perspective, the quadrant will represent non-renewable use since $t_C > t_H$, with the exception of where pumping durations are less than t_H.

d) **Non-renewable Use** (upper right quadrant): *systems where pumping is greater than the maximum capture, and also have long response times ($Q > Q_{max}$, $t_C > t_H$).* From a flux-based perspective, this is a situation of non-renewable groundwater use. From a storage-based perspective, this quadrant may nevertheless represent renewable use for the same reasons as the lower right quadrant. However, the conditions under which this is possible are much more constrained owing to the larger t_C.

Only in the bottom left quadrant is groundwater use always renewable under both flux and storage-based definitions. In this quadrant, unless the system is highly hysteretic, the recovery time will also approximately equal the time for an aquifer to reach a new dynamic equilibrium after the onset of pumping. Hence, we propose here that:

Renewable groundwater use allows for dynamically stable re-equilibrium of groundwater levels and quality on human timescales (~50-100 years).

A key implication implicit in this definition is that *defining renewable or non-renewable groundwater without considering how it is being used is impossible*. For use to be renewable it must lead to full and reversible capture of consistent quality groundwater within human timescales.

Long-term transients in groundwater levels and flows may occur well beyond human timescales due, for example, to non-stationary climate or land use change (i.e. $R_0 \neq D_0$ and $\Delta R_N \neq 0 \neq \Delta D_N$). In such a situation, from a flux perspective, the rate of pumping that can be considered renewable must also change through time. Furthermore, from a storage perspective, the concept of renewable groundwater use will only have meaning relative to the storage change that might have occurred naturally over a particular period of concern.

The bottom left quadrant is the only one that will always represent renewable groundwater use for both flux and storage/levels. If pumping is restricted in time, both the right-hand quadrants may also be considered renewable but only from a storage-based perspective. In other words, recovery of levels is possible over a human timescale after cessation of pumping but only under specific criteria particular to their quadrant i.e. depending on the particular C_{max}/Q ratio, duration of pumping and time to full capture. In these right hand quadrants, such groundwater use may still have deleterious impacts on stream flows.

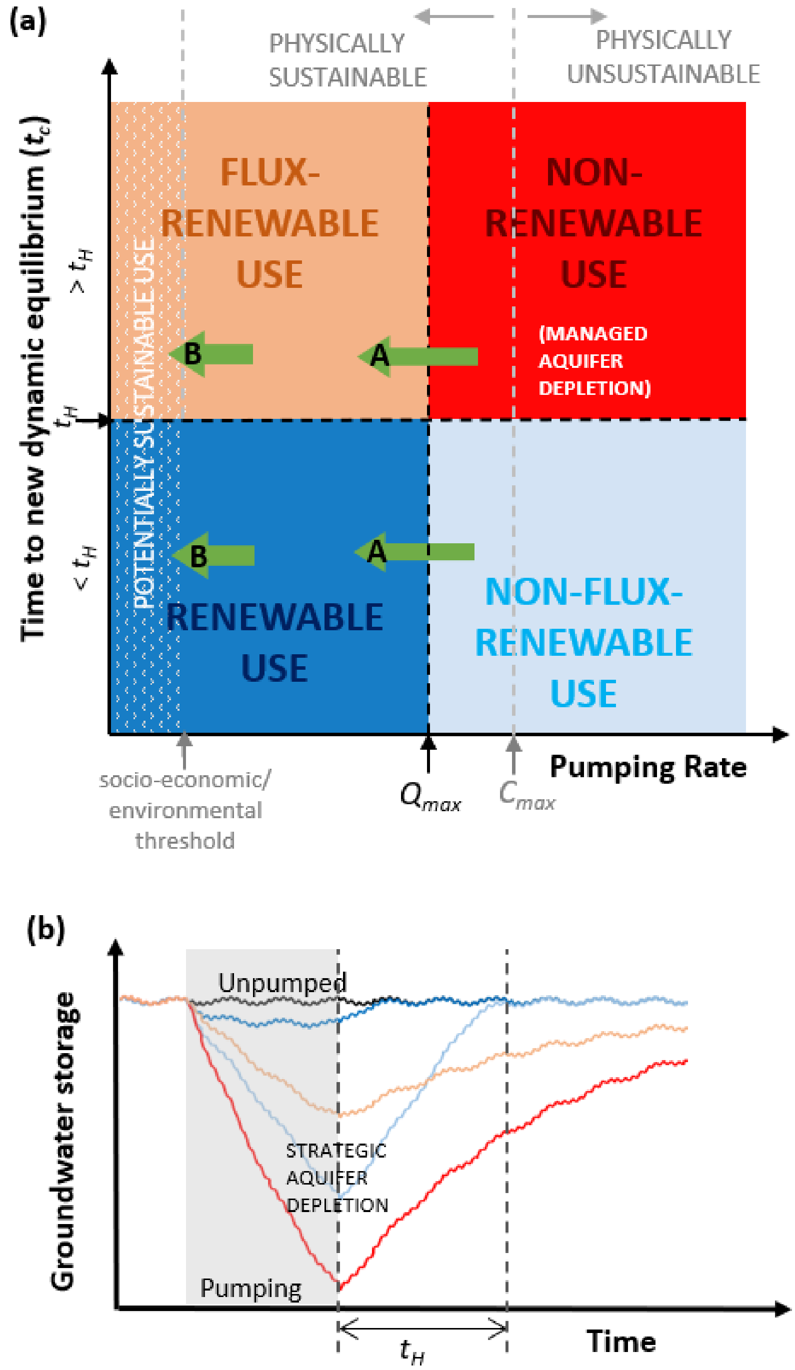

***Figure 3**. **(a)** Four quadrants of groundwater renewability and sustainable groundwater management. **(b)** time-series indicating a limited pumping and recovery scenario – colours of each time series match each quadrant colour in (a). Arrows A and B are described at the end of Section 5 as possible trajectories towards potentially sustainable use. C_{max} = physically sustainable pumping rate, Q_{max} = maximum rate of pumping which also allows the aquifer to recover flows and storage of consistent quality groundwater, t_H = human timescale.*

5. A four quadrant framework for groundwater resource management

The process of outlining a new definition of renewable groundwater use led us to delineate the four quadrants shown in Figure 3. This raises the question as to how this framework relates to the broader framework of groundwater sustainability. In particular, can sustainable groundwater use only be present in the lower left, renewable use, quadrant? To consider this, we employ the following definition of groundwater sustainability from Gleeson et al (2020):

> *"Groundwater sustainability is maintaining long-term, dynamically stable storage and flows of high-quality groundwater using inclusive, equitable, and long-term governance and management."*

Under this definition, the first criteria of '*dynamically stable storage and flows'* restricts us to the left hand quadrants where pumping is less than Q_{max}. The second criteria of '*high-quality groundwater'* is more open-ended and context specific, but as a minimum we can imagine further restrictions on the pumping rate to ensure local environmental flow and quality requirements can be met. The third criteria '*using inclusive, equitable, and long-term governance and management'* implies it is important to consider how groundwater is embedded in complex social and ecological systems which are not well represented in conventional approaches to study and promote groundwater sustainability (Huggins *et al.*, 2022). Furthermore, it could be imagined in different contexts that this determination may or may not allow for groundwater use in the upper left quadrant to be considered sustainable (Figure 3a), despite being non-renewable use from a storage perspective, depending on whether the extent of the storage depletion and associated environmental impacts are considered socially and economically acceptable.

To illustrate further this issue while describing some of the features of each quadrant, we now present a series of short case studies drawn from the literature, and show how this four-quadrant mapping may be a useful tool for delineating groundwater management pathways towards more sustainable groundwater use. We have drawn these case studies from our understanding of the literature to help illustrate and contextualise our concepts, but acknowledge the limitations of our understandings and that local conditions in any of these case studies may differ from the generalities in literature.

Lower Quadrants ($t_C < t_H$)

Inherently, within these lower quadrants with fast response times (low t_C), the pre-development rates of recharge and discharge are likely to be in balance (i.e. $R_0 \sim D_0$). Furthermore, problems due to over-abstraction may become quickly apparent. While this may be a disadvantage when climatic shocks such as droughts occur, it also means problems related to over-pumping can quickly be addressed and mitigated within human timescales. We give two contrasting groundwater-use case studies here for two different types of aquifer: crystalline basement and Chalk (limestone).

Crystalline Basement

Weathered crystalline rock aquifer systems in humid equatorial Africa are discontinuous but occur across much of the wet tropics (Taylor and Howard, 2000). They typically comprise a shallow unconfined (or semi-confined) aquifer within an unconsolidated regolith that is often hydraulically connected to an underlying aquifer within fissured bedrock (Lachassagne, Dewandel and Wyns, 2021). Seasonal (monsoonal) recharge occurs annually (Owor *et al.*, 2009; Kotchoni *et al.*, 2019), groundwater levels consistently lie above the base level of surface drainage, and groundwater drains to perennial surface waters including lakes, rivers and wetlands (Cuthbert, Taylor, *et al.*, 2019). These systems are not often thought to form large-scale regional aquifers, with smaller local flow systems more common, governed by fracture networks, lithological discontinuities and faults. Furthermore, in the humid tropics where recharge is generally high, the distance between perennial streams is often less than a few km (Cuthbert, Taylor, *et al.*, 2019). Transmissivity is typically low (<10 m^2/day), and specific yield is moderate to low (5% +/- 4%), leading to characteristic groundwater response times commonly less than 50 years. In combination with typically low pumping rates, often but not always constrained by low transmissivities, groundwater use is likely to be renewable in many parts of such aquifers, thus they are situated in the lower left quadrant.

In other parts of the world, similar crystalline basement aquifer systems with quick response times are sometimes being over-pumped with respect to the maximum capture, moving them into the lower right quadrant. Some of the hard rock aquifers in Peninsular India fall into this category, commonly being storage-limited with dense drainage networks (Fishman *et al.*, 2011; Hora, Srinivasan and Basu, 2019; Bhanja, Mukherjee and Rodell, 2020). For example, shallow, fractured hard rock aquifers with low yields in Telangana region, Andhra Pradesh are important sources of irrigation for a major rice producing area (Fishman *et al.*, 2011). Since most of these rocks have negligible primary porosity, while pumping rates may regularly exceed rates of maximum capture, the limited groundwater storage places a physical restriction on irrigation from year to year. Consequently, irrigated extents have to be reduced when the pre-season depth to groundwater is low, and water table fluctuations explain a significant part of the large annual variability in irrigated or rice cultivated areas. More broadly across Southern India, groundwater recovery has been reported based on GRACE satellites, water level measurements in groundwater monitoring wells and global hydrological models (Asoka *et al.*, 2017; Bhanja, Mukherjee and Rodell, 2020) which are at odds with reports of well failure and decreases in the land area irrigated from shallow wells (Hora, Srinivasan and Basu, 2019). Hora et al. (2019) argue that recently reported results are skewed by the problem of ‘survivor bias’, with dry or defunct wells being systematically excluded from trend analyses due to missing data and that including these wells reveals increasing groundwater stress in South India.

Chalk (limestone), northwestern Europe

The Chalk (limestone) is an important regional aquifer in north-western Europe. Its hydraulic properties derive primarily from the presence of fractures and their development through dissolution that is enhanced under phreatic (unconfined) conditions and in valley bottoms (MacDonald and Allen, 2001; Allen and Crane, 2019). Groundwater response times are generally <<50 years owing to the high transmissivity (>1000 m^2/d is common) in combination with low specific yield (< a few percent) and relatively short distances between perennial streams. Seasonal (predominantly winter) recharge (Goderniaux *et al.*, 2021) occurs annually and groundwater discharge provides baseflow to rivers and chalk streams, some of which are seasonal. Groundwater abstraction in Europe is often well-regulated to protect streamflows under the EU Water Management Directive, hence in many parts of the aquifer the licenced as well as actual use is significantly less than the maximum rate of capture. This is the case in the Paris Basin (Rouillard, 2020) for example where an agreement with farmers to restrict abstraction volumes for irrigation has been achieved (Lejars *et al.*, 2012).

However, incidences of localized intensive pumpage exceeding capture have recently been noted in the drying of chalk streams following successive dry winters in southeastern England (Jackson, Bloomfield and Mackay, 2015; Allen and Crane, 2019). Overall, the groundwater use of this aquifer is often renewable, and when the impacts of prolonged drought do manifest, the levels and flows are quick to recover once winter recharge resumes.

Upper Quadrants ($t_C > t_H$)

Inherently, within these upper quadrants with slow response times (high t_C), problems due to over-abstraction may take more than a lifetime to become apparent. While this may be an advantage when climatic shocks such as droughts occur, it also means problems related to over-pumping cannot quickly be addressed and mitigated within human timescales. We give two contrasting groundwater-use case studies here for contrasting sedimentary aquifers.

Nubian Sandstone Aquifer, northeastern Africa

An important example of a regional aquifer with a long groundwater response time, the Nubian Sandstone Aquifer, is a partly phreatic aquifer of fine to coarse grained sandstones of Late Triassic to Early Cretaceous age (Heinl & Brinkmann, 1989; Salem, 2016). Current recharge rates are thought to be low for the Nubian Sandstone Aquifer, although exactly how low is a matter of some debate (Darling *et al.*, 1987; Sultan *et al.*, 2011; Ahmed and Abdelmohsen, 2018), with little opportunity for increasing recharge due to pumping. The large response times cause long lag times between recharge and discharge such that the aquifer heads (i.e. storage) are still receding from the last major recharge period 10,000 years ago (Lloyd and Farag, 1978). Hence, while capture of discharge to oases and the seaward boundary will eventually occur, any significant groundwater development will also lead to sustained long-term aquifer depletion. In light of this, very low rates of abstraction would be required to keep this aquifer in the upper left quadrant to maintain a state of flux-renewable use. In the present state of development the rates of abstraction are much larger, meaning that 'managed aquifer depletion' is the current groundwater management situation. Owing to the combination of very high response time and major historic change in recharge rate, the concept of renewable use with respect to storage is perhaps not very meaningful in this, or similar, cases.

Permo-Triassic sandstone aquifers, UK

Alongside the Chalk aquifers discussed above, Permo-Triassic sandstone aquifers represent the most important groundwater resources in the UK. They comprise mostly aeolian, weakly cemented, red sandstones with high intergranular hydraulic conductivity enhanced by fracturing (Allen *et al.*, 1997). Owing to the moderate transmissivity (median ~200 m^2/d) in combination with high specific yield (~10% is typical) groundwater response times are expected to be >50 years where distances between discharge boundaries are large enough (Rushton, 2004). Recharge occurs predominantly in winter and groundwater abstraction is well-regulated to protect streamflows under the EU Water Management Directive. In many parts of the aquifer the licenced, as well as actual, use are significantly less than the maximum rate of capture (Morris, 2021), although over-licensing has been highlighted in some areas (Hudson, 2002). Hence these aquifers often likely straddle the upper-left and lower-left quadrants, being operated under flux-renewable use, but not always being used renewably from a storage perspective where response times are greater than human timescales.

Pathways and strategies for renewable and sustainable groundwater use

In many instances (but see next paragraph for exceptions), response times of groundwater systems may not change significantly on management timescales. As such, those aquifers situated in the upper quadrants may never be used in a renewable way from a storage perspective even if pumping was reduced to flux-renewable or even sustainable levels. Hence, pathways towards more sustainable pumping will generally represent movement on the quadrant plot from right to left rather than from upper to lower quadrants. Increasing pumping shifts aquifers from left to right on Figure 3a, but we focus here on the implications and strategies for shifting from right to left. This may occur via reducing pumping towards flux-renewable use from situations where pumping is currently greater than Q_{max} (Figure 3, A arrows). This is likely to lead to initially substantial and rapid increases in groundwater levels which may be an important economic imperative with respect to equitable access to groundwater in regions where domestic wells are at risk from drying owing to agricultural or industrial abstraction (Jasechko and Perrone, 2021). Further pumping reductions from situations of flux-renewable use towards sustainable use (Figure 3, B arrows) would enable the re-establishment of previously perennial streams and groundwater dependent ecosystems, as discussed in the Chalk example above. There are also many examples of urban aquifers around the world where post-industrial decline of abstraction or switching from groundwater to surface water sources has led to aquifer recoveries along both such pathways (Render, 1970; Buxton and Shernoff, 1995).

In some instances, there may be significant transience in a groundwater system's hydraulic response time. This can occur for a number of reasons such as aridification of a recharge area as occurred historically for the Nubian Sandstone aquifer over the last 10 ka (Abouelmagd *et al.*, 2014; Voss and Soliman, 2014), or land use change such as the clearing of native vegetation for agriculture in Niger in the more recent past which has led to increased groundwater levels and a greater degree of groundwater-surface water connectedness (Favreau *et al.*, 2009). Hence, the context for groundwater management within the quadrant diagram may not be fixed in time due to transience in such groundwater systems.

Aquifers currently in the lower right-hand quadrant, despite being in a situation of non-renewable use from a flux perspective, are sometimes managed for 'strategic aquifer depletion' (Figure 3b). This is where the aquifer is used temporarily, for example during a long drought period, at a pumping rate that is well above than $Q_{\max}$, but in the knowledge that once pumping ceases, the levels and flows will return to their previous equilibrium values within human timescales.

The High Plains Aquifer (HPA or Ogallala Aquifer), USA, usefully illustrates some of these issues. It is one of the largest aquifers in the world, comprising Neogene-age, generally poorly consolidated, gravels, sand, silt and clay (Divine, Eversoll and Howard, 2018; Bruun et al 2016) and is mostly unconfined. The central portion of the Northern HPA in Nebraska is thought to receive 100-200 mm/y of recharge (Scanlon *et al.*, 2012) and has the highest saturated thicknesses, and hence transmissivities, of the whole aquifer unit and is still well connected to the surface drainage (Scanlon *et al.*, 2012). Hence, response times may be within human timescales indicating a fully renewable case (lower left quadrant) where pumping is often less than the maximum capture rates. However, as one moves further southwards within the aquifer, recharge decreases substantially by as much as two orders of magnitude but the aquifer is still highly utilised for irrigated agriculture. Hence in the Southern Ogallala Aquifer (SOA), currently groundwater levels are, to a great extent, well below drainage base levels, and pumping rates are larger than maximum capture. While strategic depletion as part of sustainable groundwater management may be theoretically possible, in principle, in parts of the aquifer, withdrawal rates have been persistently exceeding capture for the SOA for decades. Recent efforts to reduce further depletion in the HPA have shown that if recharge and pumping are balanced after maximum capture has been exceeded, a new equilibrium can be achieved (Butler *et al.*, 2018). While this does not reverse ecological damage that has been done by lowering groundwater levels below streams, it does provide a path forward that allows for continued groundwater extraction in heavily stressed aquifers.

6. Challenges and approaches to implementing the quadrant framework

For the proposed framework to be useful, the key metrics of capture magnitudes and timescales need to be reliably estimated. Recommending specific management approaches for specific quadrants or aquifers is beyond the scope of this manuscript but herein we suggest a general approach to analysis.

A spectrum of approaches is available, of increasing detail depending on the data availability, budgetary constraints and risk-reward context of the decision making process. Space precludes a thorough review of various available methods (but for example see Li *et al.*, 2022), but developing a hydrogeological conceptual model should always be the starting point (Rushton, 2004; Rushton and Skinner, 2012), and where sufficient data are available the well-tested combination of iterative field observation and analytical or numerical modelling should be the preferred approach (Ferré, 2017).

Appropriately evaluated groundwater models with properly constrained uncertainty bounds can provide the necessary tools for testing the likely capture magnitudes and timescales of a groundwater body, and subsequent management. However, in many parts of the world, long-term groundwater monitoring data and detailed field investigations on aquifer parameters or forcing data are not currently available. Furthermore, sufficient economic or human resources may be too sparse to make this ‘gold standard’ approach viable. In such contexts implementing the quadrant framework may be challenging. Hence, we offer the following suggestions for ways to combine expert judgement with a minimum amount of data, and to guide the appropriateness of any simplifications to the water budget equations (Equations 1-4) which might make the calculations more tractable until more data and/or resources are available.

Where detailed modelling is not possible due to resource constraints, or warranted due insufficient data for forcing or evaluation, the capture timescale may be approximated via a groundwater response time (GRT) calculation (Kraijenhoff van de Leur, 1958; Alley *et al.*, 2002; Rushton, 2004; Rousseau-Gueutin *et al.*, 2013; Simpson, Jazaei and Clement, 2013; Cuthbert, Gleeson, *et al.*, 2019). However, this should be used with care, and with due attention to the likely direction and magnitude of any uncertainties involved. The distance between groundwater fed perennial streams or other hydraulic boundaries (L) is normally straightforward to approximate (Erskine and Papaioannou, 1997; Cuthbert, Gleeson, *et al.*, 2019), but since this term is squared in the GRT equation, its uncertainty should be duly incorporated. Hydraulic properties of the aquifer (S, T) under consideration may be more uncertain, depending on the available mapping and prior field investigation.

> “A responsible hydrogeologist should be aware of the theoretical aquifer response rate of *their* aquifers—particularly for ones not well known or well monitored.”
>
> Eskine & Papaioannou (1997) *edited for gender inclusivity*

The groundwater response time is then proportional to L^2S/T with the constant of proportionality to be chosen depending on the geometry of the system and the proportion of the total re-equilibrium (e.g. 63% e-folding, 90%, 99%) as appropriate.

If the system has a short groundwater response time, it is likely to be in dynamic equilibrium and it may be safe to assume $R_0 = D_0$. If no significant trends in recharge are expected due, for example, to climate or land-use changes, then it can also be assumed that $\Delta R_N \sim 0 \sim \Delta D_N$. Once full capture has been achieved, $\frac{dS}{dt} = 0$, and using Equation 1-3, the maximum capture becomes $C_{\max} = \Delta R_Q - \Delta D_Q$. Since the maximum value of ΔD_Q will equal $-D_0$ ($= -R_0$), in this case the maximum capture, $C_{\max} = R_0 + \Delta R_Q$. Hence, in some situations where the assumption $\Delta R_Q \sim 0$ is warranted, an estimate of the maximum capture may be made by assuming $C_{\max} = R_0$. In similar situations and where ΔR_Q is likely significant but unknown, R_0 will be a minimum estimate for the rate of $C_{\max}$. However, it is crucial to note that actually pumping at this rate would lead to zero groundwater discharge which would be disastrous to terrestrial and aquatic groundwater-dependent ecosystems.

This discussion raises the issue of the so-called 'water budget myth' (Bredehoeft et al. 1982) which asserts that the rate of R_0 is not of relevance to rates of physically sustainable pumping. Based on the logic of the previous paragraph it is apparent that the water budget myth is itself a myth, at least in some circumstances. Furthermore, it is of note that every paper (that we are aware of) that has weighed-in on the 'water budget myth' debate over the past 40 years has been either explicitly or implicitly predicated on the assumption of a short response time aquifer situated in a stationary recharge regime i.e. $R_0 = D_0$ and $\Delta R_N = 0 = \Delta D_N$ (Bredehoeft, et al. 1982; Bredehoeft, 2002; Devlin and Sophocleous, 2005, 2006; Kalf and Woolley, 2005; Loáiciga, 2006, 2017; Zhou, 2009). Presumably this is for didactic reasons for conceptually dealing with the principle of capture in a similar vein to our thought experiment above.

However, since it is likely that many, if not most, of the world's major aquifers actually have response times rather longer than human timescales (Bredehoeft and Durbin, 2009; Rousseau-Gueutin *et al.*, 2013; Cuthbert, Gleeson, *et al.*, 2019), this is a fundamental oversight. Theoretically at least, for periodic recharge forcings, R_0 and D_0 are likely to be out of phase as long as GRT is greater than the period of the forcing (Townley, 1995). The lag time between a recent step change in R and the subsequent impact on D, will also approximate the GRT (Rousseau-Gueutin *et al.*, 2013). Furthermore, it shouldn't be automatically assumed that recharge is stationary on human timescales owing to climate variability (both natural and anthropogenically altered) or land-use change (Favreau *et al.*, 2009).

Hence for large response time systems, there may be transients in the long term water balance, R_0 and D_0 may be in disequilibrium, and it should not be assumed that $R_0 = D_0$. Neither should it be assumed that it is only pumping which could cause a long term change in R and D since these fluxes may effectively always be in disequilibrium i.e. that $\Delta R_N \neq 0 \neq \Delta D_N$. However, very large GRT systems tend to be more prevalent in more arid parts of the world where it may be safe to assume less feedback between recharge and heads in the aquifer. In these contexts it may be reasonable to assume that $\Delta R_Q \sim 0$, but we note that recent research indicates this may not be as safe an approximation than is often presumed (Quichimbo, Singer and Cuthbert, 2020).

These considerations indicate that, for many aquifers around the world, transients in various water balance elements may be important to consider and therefore the long term influence of widespread pumping may be hard to ascertain clearly amidst these natural transients from monitoring alone. Hence we maintain here that the key to robustly understanding capture is by a combination of long term monitoring and bespoke numerical modelling, in the context of environmental change. This is, after all, one of the main reasons 'why hydrogeologists model' (Bredehoeft, 2002). In this context, the advice of Bredehoeft & Durbin (2009) is apposite:

> *"…it takes some ground water systems an inordinately long period to reach a new equilibrium. The time may be so long that the fact that a new equilibrium eventually is reached becomes meaningless. The bottom line is—it is important to predict the time trajectory of ground water systems, especially if one hopes to manage the system."*

With respect to collecting the critically important long-term groundwater monitoring for good groundwater management, data not just of levels, flows and groundwater quality are needed, but also improved ways of recording or estimating variations in groundwater abstractions at large scales (Butler Jr. *et al.*, 2021).

Even if the hydraulic response times and maximum capture rates can be estimated with sufficient certainty using a relatively simple approach, including the water quality aspects in these calculations may be more challenging. In most contexts as stated above, we anticipate that the time taken for water quality to recover to its pre-pumping state may be much higher than the hydraulic response time, and the rate of maximum quality capture ($Q_{\max}$) may be significantly lower than the physically sustainable pumping rate ($C_{\max}$). This is a key challenge for the application of the framework and the importance of a sound conceptual hydrogeological model is again paramount, so that the importance of such considerations can be anticipated.

7. Having "renewed" the discussion, how can we sustain it?

This commentary was motivated by our collective perception of inconsistency in the meaning and use of the terms 'renewable groundwater' and 'groundwater renewability'. Clarification of these terms is surely vital in the quest to improve sustainable management of groundwater, by enabling identification of clear goals and pathways to that end, and aiding clear and consistent communication between scientists, managers, policy makers and stakeholders.

We began by questioning in what way groundwater can be considered a renewable resource and concluded that it can't be defined as such without reference to the manner in which it is being used. We proposed a new definition: Renewable groundwater use allows for dynamically stable re-equilibrium of groundwater levels and quality on human timescales (~50-100 years). We then showed how this leads to a useful four quadrant framework for quantitative groundwater resource management and illustrate this with reference to case studies from aquifers around the world.

A logical next step is to extend and apply our framework spatially towards realising the water management typology called for by (Foster and MacDonald, 2014).

"By understanding the aquifer characteristics relevant to water security, it should be possible to establish aquifer typologies which will respond in a similar manner to external stresses—either from humankind or climate. These typologies could then be mapped along with current groundwater status, permitting clearer communication on the groundwater dimensions of water security at the political level."

As with any overarching framework, important local details cannot be resolved at the same time as taking a panoramic view. However, we realise that for robust groundwater management such details will need to be brought into clear focus to give the necessary contextualisation in any given environment.

Aspirationally, we hope that increasing experience of recovering, previously highly-stressed, aquifer systems will aid the conversation (Butler Jr. *et al.*, 2021). Within the urban context, useful lessons can also no doubt be learned from recovering groundwater systems in many post-industrial cities around the world (Lerner and Barrett, 1996).

We hope this paper will stimulate practical reflections and therefore we end with the following questions, as a way to promote further and wider dialogue on this issue, and to encourage the reader to reflect on the practical possibilities of groundwater resources in their own experience:

- *Where do aquifers of concern currently plot on the quadrant framework?*
- *Where would stakeholders, managers and decision makers like them to plot?*
- *What actions can be taken to get them there?*

We leave it to the community to explore the utility of the framework we have proposed here and continue the discussion. We hope that sustaining such conversations can yield safer, more renewable and less stressfully depleting definitions of groundwater use.

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